Fire Ecology and Fire Effects

Integrating Fuel Treatments into Comprehensive Ecosystem Management

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Abstract—*To plan fuel treatments in the context of comprehensive ecosystem management, forest managers must meet multiple-use and environmental objectives, address administrative and budget constraints, and reconcile performance measures from multiple policy directives. We demonstrate a multiple criteria approach to measuring success of fuel treatments used in the Butte North Strategic Placement of Treatments (SPOT) pilot project. Located in the Beaverhead – Deerlodge National Forests, Montana, the project addresses multiple issues: altered wildlife habitat affecting sensitive species, grassland conversion to forest, an insect epidemic, water resource concerns, wildland-urban interface development, and wildland fire management. Managers are working with researchers to develop dynamic landscape management strategies. They employ multiple modeling approaches to conduct an integrated assessment of ecological and resource issues relative to multiple management scenarios. Besides evaluating effects of proposed treatments on changes to fire behavior, they also evaluate effects on wildlife habitat, disturbance processes, water quality and economics of treatment alternatives. The intent is to effectively integrate fuel management with Forest Plan goals and comprehensive ecosystem management. This approach offers a structure to use multiple criteria to evaluate success of fuel management activities in the context of other resource objectives.*

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Introduction

Recent dramatic increases in wildland fires triggered the commitment of substantial resources to reduce hazardous fuels. The Government Accounting Office (2002) calls for federal land management agencies to develop "consistent criteria to identify and prioritize" areas requiring treatment and "clearly defined outcome-oriented goals and objectives." The urgency to reduce forest fuels creates tension with expectations that forest management must address competing resource objectives while applying the best available ecosystem science. The Healthy Forest Restoration Act of 2003 established a framework to conduct hazardous fuels reduction projects on federal forested lands to protect key ecosystem components, reduce risk to communities and municipal water supplies, improve critical habitat for threatened or endangered species, restore vegetation structure to reflect historic variability, improve commercial value of forest biomass, and address insect infestation. How do managers effectively integrate the complexities of ecosystem science and multiple resource objectives into practical planning strategies?

The scientific basis for comprehensive ecosystem assessment is well established (Grumbine, 1997) and issues of applied ecosystem assessment have been thoroughly discussed (Haynes et al. 1996; Holt 2001; Jakeman and

Letcher 2003; van der Sluijs 2002). Provisions for conducting environmental impact analysis and managing resources to meet multiple objectives were established in the National Environmental Policy Act of 1969 and National Forest Management Act of 1976, respectively.

Computer-based decision support systems evolved concurrently with ecosystem sciences. Numerous modeling systems seek to transfer ecosystem theory and knowledge into practical management solutions. Many modeling tools focus on resource specific issues such as water quality, wildlife habitat, wildland fire behavior, vegetation processes, management logistics, and economic resource assessment. Many modeling tools coevolved with geographic information systems (GIS) permitting spatially explicit model displays. The need to assess integrated ecosystem components drives development of the emerging field of Integrated Assessment Modeling (IAM) (Jakeman and Letcher 2003; van der Sluijs 2002). In principle, IAM accounts for ecological, social, and economic values where planning environmental and resource management activities. The objective of IAM is to integrate multiple, relevant modeling components into a unified framework to improve how complex environmental problems are analyzed and possible solutions identified.

This paper presents a conceptual framework for a modeling-based assessment and planning procedure that integrates forest fuel treatments with multiple resource objectives. The framework is an example of an IAM currently used for the Butte North Project, Beaverhead-Deerlodge National Forest, Montana. The project is as a pilot of the USDA Forest Service, Strategic Placement of Fuels (SPOT)program. The SPOT program is intended to guide development of a "consistent and systematic interagency approach" to identify and plan treatments on forested acres deemed most critically in need of fuel reduction (Bosworth 2005). The framework is presented in a structured, stepwise format, and provides insight into how integrated assessment modeling is practically implemented. We conclude by describing a "performance report card" for evaluating treatment success based upon multiple resource objectives.

Study Area

The Butte North Project area, located in Silver Bow County, Montana, covers 38,600 ac, 80% of which is managed by the Beaverhead-Deerlodge National Forest (BDNF) (figure 1). In the lower elevations, shallow, highly erodible soils support grass and sagebrush lands. The forested lands above are dominated by lodgepole pine (*Pinus contorta*) with 2,800 ac of Douglas-fir (*Pseudotsuga menziesii*) in drier sites. The area was heavily impacted by mining throughout the late 19th and early 20th century (Lyden 1948). Most of the timber was removed to support mining operations. Commercial logging of lodgepole pine occurred most recently during the 1980's. Many forest roads intersect stream channels. Over 80 residential structures occupy the wildland-urban interface. Small ranch operations run cattle on private lands and federal grazing allotments. The National Forest lands are highly valued for hunting and other recreation. A small municipal water supply reservoir is also located within the project area.

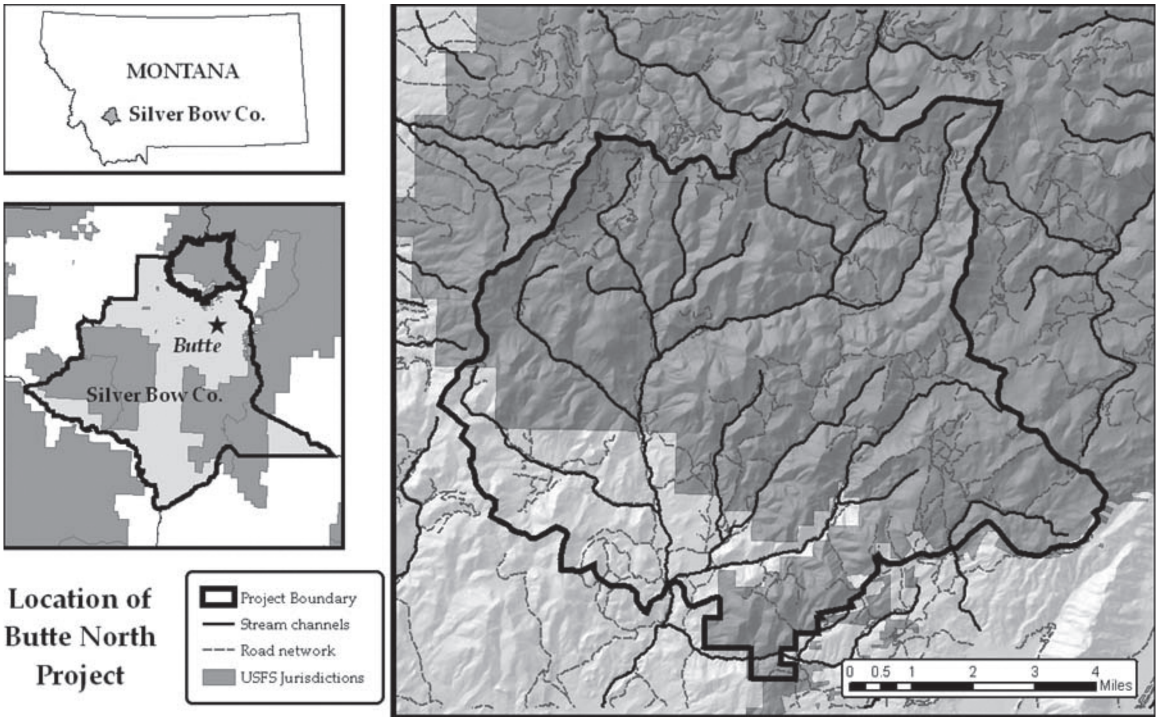

Figure 1—Location of study area within Silver Bow County, Montana.

Current Conditions and Management Issues

The land use history and current environmental conditions result in multiple management issues. Details follow by seven general resource topics as defined by the BDNF managers. These topics are repeated in major sections of the paper as we describe the integrated modeling process.

A. Vegetation: Dense seedling and sapling cohorts occupy stands commercially harvested 20-30 years ago. Conifers continue to encroach upon grass and sagebrush lands. Understory development within Douglas-fir stands increases acres of densely stocked, multi-story vegetation. There are few stands of large mature trees, limiting the potential development of more complex 'old-growth' type vegetation structure. Encroachment and increased vegetation density generally reduces landscape complexity.

B. Insects: Infestations of mountain pine beetles are present and threaten to spread rapidly throughout the conifer forests causing extensive mortality to lodgepole and Douglas-fir stands.

C. Fire and forest fuels: Continuous stands with heavy fuel loading could provide conditions for rapid fire growth. Vegetation on over half of the managed area is classified as Fire Regime Condition Class 3 (FRCC3), indicating that conditions are departed from the historic range of variability and that significant management may be needed for restoration (Hann and Strohm 2003). Fuel loadings in beetle infested areas may increase in the future as infested trees senesce.

D. Watershed: Stream channels are over-widened and contain uncharacteristic volumes of fine sediments, probably from past mining activities and the extensive forest road network. Willow is regenerating poorly, in part due to conifer encroachment and over-grazing in riparian zones.

E. Wildlife habitat: The trend toward lower vegetation complexity probably limits habitat for species which historically inhabited the area. Plans for any proposed management activities must consider habitat for multiple aquatic and terrestrial sensitive species including red squirrel, *Tamiasciurus hudsonicus* (nesting, foraging), lynx, *Lynx canadensis* (den, foraging), black-backed woodpecker, *Picoides arcticus* (habitat), pileated woodpecker, *Dryocopus pileatus* (nesting, foraging), flammulated owl, *Otus flammeolus* (nesting, foraging), northern goshawk, *Accipiter gentilis* (nesting, foraging), fisher, *Martes pennanti* (den, foraging) and West Slope Cutthroat Trout (*Oncorhynchus clarki lewisi*).

F. Social: Dense fuel concentrations proximate to residential structures and within the municipal watershed could threaten lives, property, and a drinking water source should severe wildland fire occur.

G. Economics: Funds to conduct any management activities are limited. Proposed activities must be logistically and economically feasible.

Developing an Integrated Modeling Framework

The core Butte North assessment team consisted of specialists in silviculture, wildlife, GIS, fire and fuel management, hydrology, fisheries, and landscape modeling. Following background research, group discussions, and field reconnaissance, the team defined resource issues and developed a list of possible management objectives. The objectives were translated into landscape components and relationships that could be defined within a GIS and modeling applications. Rules were developed to adapt these components and relationships into assessment logic within the modeling framework. Modeling tools appropriate to resource issues were implemented addressing vegetation, insect spread, fuels and fire, wildlife habitat, and human uses. Modeling results were integrated into a final modeling system which assessed the feasibility and trade-offs associated with multiple objective scenarios. In summary, the IAM process was accomplished through the following steps:

Step 1: Translate Issues to Objectives
Step 2: Translate Objectives to Modeling Logic
Step 3: Build and Integrate Models
Step 4: Define Basis for Scenario Comparison
Step 5: Frame Alternative Scenarios

The IAM process permits visualization of possible consequences of multiple plausible alternatives which may help estimate and confirm anticipated benefits and conflicts. IAM may also reveal unanticipated opportunities and pitfalls. The intent is to provide spatially explicit comparison across a range of alternative scenarios.

Step 1: Translate Issues to Objectives

The core team developed a series of management objectives defined by specific activities, to address the seven identified landscape issues.

A. Vegetation: Implement pre-commercial thinning in stands commercially harvested over the past 2-3 decades. Restore grass and sagebrush lands using slashing and broadcast burning. Reduce Douglas-fir understory vegetation. Protect selected stands with larger stem sizes, passively managing for potential 'old growth' conditions. Monitor spatial arrangement of vegetation activities for changes to the mosaic of vegetation structure.

B. Insects: Thin beetle infested stands to reduce competition among the remaining trees and salvage value of some trees in infested areas.

C. Fire and forest fuels: Reduce forest fuels within stands with highest potential for extreme fire behavior. Reduce vegetation density in FRCC3 areas. Reduce vegetation density in beetle infested areas.

D. Watershed: Limit or prohibit management activities near stream channels, especially where sensitive species are present. Remove conifers encroaching into broadleaf riparian vegetation.

E. Wildlife habitat: Monitor and constrain management activities which alter potential habitat for species of concern. Minimize impacts to currently suitable habitat and favor change which increases suitable habitat.

F. Social: Reduce loading of forest fuel near structures and within the municipal water supply watershed.

G. Economics: Use commercial values from vegetation treatments which yield merchantable timber to generate revenues to fund other, non-commercial resource improvements.

Many of these objectives could be addressed simultaneously through activities within the same landscape area. For example, revenues from harvesting to reduce stand density within insect infested areas could help fund stream restoration projects. Conversely, activities to meet one objective could directly conflict with other resource objectives. For example, mechanical activity to reduce forest fuels could increase sedimentation to streams and alter sensitive wildlife habitat. The challenge of the IAM approach is to define resource relationships sufficiently well to illuminate benefits, trade-offs, and conflicts within the modeling environment.

Step 2: Translate Objectives to Modeling Logic

With objectives defined, the next step was to determine which resource components to model and to identify available data. Each objective was reviewed to determine which physical and landscape attributes best describe the features affected by the objective and how these features relate to the planning landscape. Implicit in these definitions is the requirement that spatial data be available. This is an iterative process which requires dealing with "chicken or egg" logic; prior knowledge of model input requirements may limit data that can be used, while available data may limit which modeling tools may be used (Mulligan and Wainwright 2004). Also, available data may not be sufficient; more data may need to be collected, parameters may need to be estimated from existing data, or alternative modeling approaches may be necessary.

The *minimum modeling unit*, the smallest land area identified as having unique characteristics, was also chosen at this step. The convention defining vegetation stands (hereafter "stands") as a minimum mapping unit logically translated to the minimum modeling unit. All computations and summaries are based upon the attributes of the minimum modeling unit. Attributes were assigned to stands as a single assignment assuming homogeneity for the entire unit or as a percentage of land area occupied by a given feature within the unit. An example of percentage is the portion of a vegetation stand occupied by a stream buffer. The stream buffer is also an example of a management *zone*. Zones may define common jurisdictions, areas with common management objectives, or other classifier useful for planning and analysis.

A. Vegetation: The GIS stands layer which established the minimum modeling unit was a composite of legacy Timber Stand Management Record System (TSMRS) with vegetation updates from Satellite Imagery Land Classification (SILC) data (Redmond and Ma 1996). Each stand was assigned a dominant plant/tree species, vegetation structure class, canopy density class, and habitat type.

B. Insects: The 2005 Aerial Detection Survey (ADS) GIS layer was used to identify stands and label with current beetle infestation (USDA Forest Service 2005).

C. Fire and forest fuels: In addition to assigning FRCC classifications a fire and fuels specialist used expert opinion to translate vegetation data into definitions of fuel characteristics required for fire behavior modeling. Topographic information required for fire behavior modeling was acquired from a digital elevation model and historical weather data was acquired from a nearby weather station.

D. Watershed: Stream buffers were delineated around perennial stream channels after the Inland Native Fish Strategy (INFISH) (USDA Forest Service 2006) guidelines. A riparian recovery zone was established at 50 ft and an activities monitoring/exclusion zone was established at 300 ft. The coincidence of the 300 ft zone was appended to the stands layer as a binary attribute and the portion of a stand occupied by the riparian buffer was assigned to each stand. Areas previously identified as high priority for recovery were assigned as a priority zone.

E. Wildlife habitat: Wildlife habitat modeling required vegetation characteristics acquired from the GIS stand layer.

F. Social: The locations of structures were approximated using the Montana parcel GIS layer (available at: http://nris.state.mt.us/nsdi/cadastral/) to generate a point layer representing building clusters. Points from the GIS were adjusted to match recent aerial photos provided by the BDNF. Stands within the municipal supply watershed were attributed based on a GIS layer provided by the BDNF.

G. Economics: Activity cost estimates were provided by the BDNF. Revenue estimates from potential commercial sales were derived from the transaction evidence appraisal (TEA) procedures of USDA Forest Service Region 1 (2005), explained further in the next section. Estimates of potential harvest volumes were derived from the basic vegetation attributes of the stands layer.

Step 3: Build and Integrate Models

The data describing landscape attributes and management effects were loaded into individual resource models, or sub-models. Using independent sub-models maintains model integrity, greater process transparency, and better description of errors and uncertainties inherent in all environmental modeling (Beven 2006; van der Sluijs 2002). Sub-models may be sophisticated computer programs or very simple rules developed from research or expert opinion. Respective model outputs were organized back into the base GIS and finally compiled into a final Integrated Assessment Model.

A. Vegetation—Successional pathways: Logic for successional pathways following disturbance and management activities was adopted as previously developed from research literature and expert opinion (Chew et al. 2004).

B. Insects—Infestation spread model: Based on current conditions defined by the ADS, the projected spread of the infestation was modeled using a GIS-based approach (Shore and Safranyik 1992) adapted to fit available data. Results of the insect spread modeling were used to construct a future landscape used in the fire behavior modeling to estimate fire behavior 20-30 years in the future assuming increased insect spread and increased fuel loading as dead and dying trees senesce.

C. Fire and forest fuels: Potential fire behavior was modeled using the Treatment Optimization Model (TOM) within the FLAMMAP modeling system (Finney 2002). TOM uses GIS data layers to analyze fire spread behavior assuming fixed ignition sources, and weather and wind conditions. The resulting map suggests the location, orientation, and size of fuel treatment polygons, or TOM polygons, which may most effectively and efficiently change large fire growth. Separate TOM runs were completed using 97-99th percentile weather conditions, prevailing winds from two directions, NW and SW, and two vegetation conditions, current and future bug-infested conditions created by the insect spread model. The GIS stands were attributed to indicate coincidence with TOM polygon.

D. Watershed—Specialist analysis: Watershed analysis was limited to specialist field assessments and GIS attribution of stream buffer zones previously described.

E. Wildlife Habitat—Model of wildlife habitat zones: Wildlife zones were determined by matching GIS vegetation data with the habitat requirements of the species (Hart et al. 1998; Pilliod 2005; Ruediger et al. 2000; Samson 2005). The zones were categorized on a 0-3 scale for habitat quality and the GIS stands were attributed with the suitability rank for each wildlife zone. The wildlife zones values were summed for an overall wildlife habitat quality index.

F. Social model: The wildland urban interface (WUI) was modeled by generating a buffer extending ½ mi from each building cluster point. Stands intersected by this buffer were assigned the WUI zone attribute.

G. Economic model: Timber value was estimated by the TEA method which predicts stumpage value adjusted for sale characteristics and market indicators. Polygons in the GIS vegetation layer were assigned a mechanical treatment method based on proximity to an existing road and mean slope within the polygon; this attribute adjusts the TEA values on a stand by stand basis. Estimates of forest product volumes from mechanical activities were derived by using Forest Inventory and Analysis (FIA) data in the Forest Vegetation Simulator model (FVS) (Dixon 2002) and the Fire and Fuels Extension of FVS (Reinhardt 2003). The modeling results were compiled into a "look-up" table which associates volume estimates from activities with the antecedent vegetation.

Model Integration—Results from each sub-model were compiled first in GIS then into a master IAM system called Multiple-resource Analysis and Geographic Information System (MAGIS). MAGIS is an optimization model designed to solve complex spatial and temporal scheduling problems in natural resource management (Zuuring et al. 1995). The MAGIS modeling system is based on mixed-integer mathematical programming that includes vegetation management and an optional roads component for analyzing access and associated costs and resource impacts (Weintraub et al. 1994). Generally, if a resource can be defined in a GIS and with rules relating the resource to management effects, the resource can be accounted for in MAGIS.

Figure 2 presents a schematic of the model integration structure. The MAGIS model was prepared for sub-model data by defining the attributes to import from the GIS layers. Other definitions were entered for management activities, costs, and rules for vegetation succession, activity outputs, and management activities. *Management regimes* were defined consisting of activities, alone or in series that could be applied to accomplish project objectives. Examples included slashing and broadcast burning to restore grass and sagebrush lands and mechanical thinning in the commercial management zones. With all definitions entered, the attributed GIS vegetation layer was imported to MAGIS.

Step 4: Define Basis for Scenario Comparison

The final step for building an integrated model was to define e*ffects functions*. These establish resource characteristics to be monitored and compared between alternative management scenarios run in MAGIS. These are constructed so that the output of each effects function specifically relates to a project objective. Effects functions commonly summarize acres affected by management actions. They may be viewed as an *accomplishment* meeting an objective (e.g. sum of stream project acres treated), or an *indicator* to be monitored or perhaps constrained (e.g. change in wildlife habitat index or number of acres impacted within the 300 ft stream buffer). Virtually any number of effects functions can be defined limited by project objectives and common sense. Effects functions defined for the Butte North Project include:

A. Vegetation
- Acres of lodgepole plantation thinned (accomplishment)
- Acres of grass/sagebrush restoration candidates treated (accomplishment)
- Acres of multi-story Douglas-fir treated (accomplishment)
- Acres of potential old growth affected (indicator)

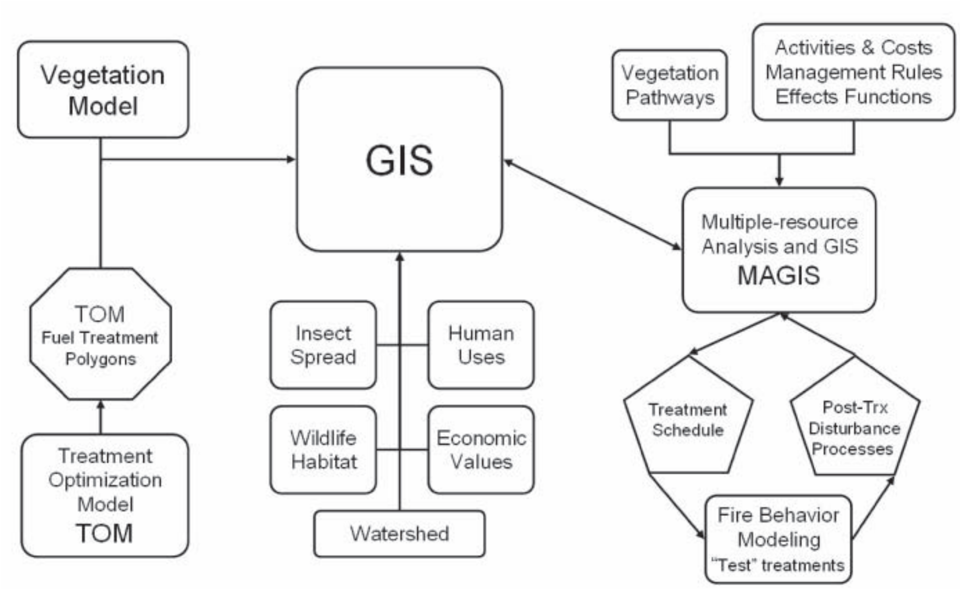

Figure 2—Schematic of model relationships and integration structure.

B. Insects
- Acres treated intersected by TOM polygons in areas of projected insect spread (accomplishment)

C. Fire and fuels
- Acres treated intersected by TOM given modeled fire behavior based on current vegetation (accomplishment)
- Acres treated classified as fire regime condition class 3 (accomplishment)

D. Watershed
- Acres of priority riparian project treated (accomplishment)
- Acres of stands treated containing any 300 ft stream buffer (indicator)

E. Wildlife habitat
- Acres treated containing habitat of key species (accomplishment or indicator depending upon associated affects)
- Index of wildlife habitat value (indicator)

F. Social
- Acres treated containing WUI buffer (accomplishment)
- Acres treated around reservoir (accomplishment)

G. Economics: These effects functions are either accomplishments or indicators depending upon other associated resource effects
- Total costs of activities
- Total product volume
- Total present net revenue

Step 5: Frame Alternative Scenarios

The process of using IAM to define alternative scenarios is similar to developing alternative land management proposals. Different combinations of desired outcomes are compiled, each emphasizing a particular set of resource objectives. A primary scenario goal or *objective function* is determined. Boolean logic is then applied to effects functions to set specific goals and apply constraints. For example, an objective function might be to maximize acres of WUI treated to reduce fuels. Constraints might be set to simultaneously limit impact in the stream protection zone, acres of mechanical treatment in the WUI zone, and budget. The mathematical solver in MAGIS first determines the feasibility of meeting the objective function within the constraints set and then calculates related impacts and outcomes defined by each effects function. Defining scenarios is an iterative and cumulative process. Results from one scenario are analyzed, adjusted, and fed into the next. This process continues until the users believe they have reached an optimal spatial and temporal schedule of treatments to meet objectives. Work on the Butte North modeling continues. Examples of basic scenarios which will be used for the Butte North analysis will include a fire threat reduction option, a wildlife option, and an economic option.

Forest Health Restoration Report Card

The IAM outlined for the Butte North Project demonstrates application of multiple modeling tools for multi-objective, multi-resource analysis. The single issue of fuel reduction does not drive the analysis. Fuels and fire threats are addressed in the context of the other significant environmental and management concerns. The opening assessment question is not, “What is the problem fire?" Instead this approach asks, "What role does fire play as one component of a complex system?" and "What management actions are warranted to address overall forest health?"

Expecting that management accomplishments must be accounted for based on standard performance criteria, the systematic assessment of key resources through the preceding analysis presents a logical foundation for a multiple criteria performance reporting tool. Given that fire and forest fuel will drive budgets for the foreseeable future and that the Healthy Forest Restoration Act establishes the management directives, the prospective tool is entitled: Forest Health Restoration Report Card. Figure 3 presents a working draft concept. The intent is to account for and acknowledge multiple costs and benefits from management activities, to concisely report expected treatments objectives, and to convey this information simultaneously to several audi-

PROJECT NAME: Butte North
LOCATION: Beaverhead-Deerlodge National Forest, Silver Bow Co., MT
PARTNERS: BDNF, MT DNRC
PROJECT SUMMARY:

			Treatment Method				Expected Treatment Effectiveness (yrs)
TREATMENT GOALS	Total	%	RX Fire Ac	%	Mechanical	%	
ACRES TREATED	**1000**	**100%**	**650**	**65%**	**350**	**35%**	
RESOURCE TOPICS							
VEGETATION							
Grass/sage restoration							
DF understory thin							
INSECTS & DISEASE							
FUEL REDUCTION							
FRCC Change	500	50%	250	25%	250	25%	
WATERSHED							
WILDLIFE HABITAT							
SOCIAL VALUES							
Wildland-Urban Interface	350	35%	200	20%	150	15%	20
Water Supply	200	20%	150	15%	25	2.5%	35
WATERSHED	250	25%	50	20%	250	100%	35
TES	50	5%	25	50%	10	20%	15
OLD GROWTH	100	10%	100	100%	65	65%	85
BIOMASS REMOVAL							
FINANCIAL ANALYSIS							
TREATMENT COST	$ (152,500)		$ (97,000)		$ (55,500)		
PRODUCT REVENUE	$89,275		$ 0		$89,275		
NET VALUE	$(63,225)				$33,775		
ECONOMIC IMPACTS							
DIRECT ECONOMIC EMPLOYMENT IMPACTS							
DIRECT ECONOMIC INCOME IMPACTS							

Figure 3—Working prototype for a Forest Health Restoration Report Card. Some cells are intentionally left empty to reflect how the single card can capture the unique character of each project.

ences. The report card should directly reflect the project purpose and need. It should document the expected resource effects, both positive and negative, expected duration of treatment effectiveness, the economic benefits and costs, and any other social effects that have been analyzed. The tool provides a valuable qualitative and quantitative summary of project goals, merits, impacts, and costs; accounts for annual accomplishments comparing treatment targets to actual acres treated; and provides a basis for future project monitoring and outcome-based performance reporting. This tool sets the foundation for measuring success beyond simply reporting acres treated and more robustly captures the value and intent of undertaking fuel and forest restoration treatments.

The report card system may be one tool to help restore public trust, because it clearly demonstrates that multiple resource and environmental concerns were addressed and acted upon. Furthermore, the report card system may provide a basis for more consistent multi-objective planning and monitoring of future projects with a forest health emphasis. Modeling results may be validated and the degree to which intentions are realized is transparent.

Future of Modeling and Performance Measures

Models may help guide decisions, not make them. Models are limited by errors and uncertainty and, as such, are never a substitute for professional judgment and ground verification of planning data. For all the error and uncertainties within the models and modeling processes themselves, we cannot hold off decisions until we have perfect systems. Models provide some measure of simplicity with the hope of greater clarity as we wrestle with inherently and intractably complex systems. Reasonably enough, management of complex systems requires tools that adequately represent this complexity. IAM is one such tool. Our current abilities to integrate resource modeling systems are coarse but will only improve with practice (Jakeman and Letcher 2003) and development of improved IAM tools and logic.

We have outlined a practical procedure for integrating fuel treatments into comprehensive ecosystem management through integrated assessment modeling. This framework provides a tool for systematic analysis of multiple resource objectives within a common planning area. Rather than fire and fuels issues driving the process, this framework provides insight into the relationship between fire, forest fuels, and other resources. The results from this integrated assessment modeling approach offer a structure to develop a multi-criteria performance report card. The outcome may be planning protocols that make better use of ecosystem science and more defensibly meet land management directives.

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Comparison of the Sensitivity of Landscape-Fire-Succession Models to Variation in Terrain, Fuel Pattern, Climate and Weather

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Abstract*—The relative importance of variables in determining area burned is an important management consideration although gaining insights from existing empirical data has proven difficult. The purpose of this study was to compare the sensitivity of modeled area burned to environmental factors across a range of independently-developed landscape-fire-succession models. The sensitivity of area burned to variation in four factors, namely terrain (flat, undulating and mountainous), fuel pattern (finely and coarsely clumped), climate (observed, warmer & wetter, and warmer & drier) and weather (year-to-year variability) was determined for four existing landscape-fire-succession models (EMBYR, FIRESCAPE, LANDSUM, and SEM-LAND) and a new model implemented in the LAMOS modelling shell (LAMOS(DS)). Sensitivity was measured as the variance in area burned explained by each of the four factors, and all of the interactions amongst them, in a standard generalised linear modelling analysis. Modeled area burned was most sensitive to climate and variation in weather, with four models sensitive to each of these factors and three models sensitive to their interaction. Models generally exhibited a trend of increasing area burned from observed, through warmer and wetter, to warmer and drier climates. Area burned was sensitive to terrain for FIRESCAPE and fuel pattern for EMBYR. These results demonstrate that the models are generally more sensitive to variation in climate and weather as compared with terrain complexity and fuel pattern, although the sensitivity to these latter factors in a small number of models demonstrates the importance of representing key processes. Our results have implications for representing fire in higher-order models like Dynamic Global Vegetation Models (DGVMs)*

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Introduction

Wildland fire is a major disturbance in most ecosystems worldwide (Crutzen and Goldammer 1993). Fire interacts with weather and vegetation such that forested landscapes may burn quickly whenever fuels are abundant, dry and spatially continuous, especially if there is a strong surface wind (McArthur 1967; Rothermel 1972). The relative importance of variables in determining area burned is an important management consideration although gaining insights from existing empirical data has proven difficult.

Landscape-fire-succession models, that simulate the linked processes of fire and vegetation development in a spatial domain, are one of the few tools that can be used to explore the interaction of fire, weather and vegetation over long time scales. There is a diverse set of approaches to predicting fire regimes and vegetation dynamics over long time scales, due in large part to the variety of landscapes, fuels and climatic patterns that foster frequent forest fires (Swanson and others 1997; Lertzman and others 1998), and variation in modeler's approaches to representing them in models.

Systematic comparisons among models, using a standardised experimental design, offers insight into our understanding of the key processes and parameters affecting diverse ecosystems (Dale and others 1985; Rose and others 1991; Gardner and others 1996; VEMAP 1996; Pan and others 1998; Cramer et al 1999) as well as our confidence in the reliability of model predictions (Bugmann and others 1996; Turner and others 1989). The objective of this research is to compare a range of landscape-fire succession models to gain insight into the relative importance of terrain, fuel pattern, weather and climate in determining modeled area burned, and the extent to which findings can be generalized across a range of ecosystem types.

We selected a set of landscape-fire-succession models and performed a comparison on neutral landscapes to identify the relative importance and sensitivity of simulated fire to terrain, fuel pattern, weather and climate. We originally planned to compare results of models from the twelve classification categories of landscape-fire-succession models of Keane and others (2004) but in reality we limited ourselves to models from three classification categories selected from modelers with the time and resources to undertake the complex simulation design. We compared five models including EMBYR (Gardner and others 1996), FIRESCAPE (Cary & Banks 1999), LANDSUM (Keane and others 2002), SEMLAND (Li 2000), and a new application of the LAMOS modelling shell (Lavorel and others 2000). These models may appear functionally similar but they are quite different in many aspects, including a wide diversity in the simulation of fire spread and ignition, representation of vegetation, and the complexity of climate and fire linkages (Cary and others 2006).

This study does not represent an exercise in model validation. Rather, we selected models that have previously been verified and validated, and one new model, and analysed their behaviour with respect to variation in terrain, fuel pattern, weather and climate. A more comprehensive description of the study is given by Cary and others (2006).

The Models

EMBYR is an event-driven, grid-based simulation model of fire ignition and spread designed to represent the landscapes and fire regimes of Yellowstone National Park (Hargrove and others 2000). The pattern of forest succession of lodgepole pine forests is simulated by a Markov model, with fuels sufficient to sustain crown fires developing as a function of forest stand age. The probability of fire spreading from a burning pixel to each of its neighbors is determined by stand age, fuel moisture, wind speed and direction, and slope. An index of fire severity, based on fuel type, fuel moisture, wind speed and the rate that the cell burned, determines whether fire intensity is sufficiently high to cause a stand-replacing fire.

FIRESCAPE simulates individual fire events that are combined into patterns of fire frequency, fire intensity and season of occurrence (Cary and Banks 1999). Daily weather is generated by a modified version of the Richardson-type stochastic climate generator (Richardson 1981) so that serial correlations within a particular meteorological variable and cross correlations between variables are maintained (Matalas 1967). Ignition locations are generated from an empirical model of lightning strike modified from McRae (1992).

The rate of spread of fire from a burning pixel to its neighbors is assumed to be elliptical (Van Wagner 1969) and is determined by Huygens' Principle, although varying topography, fuel load and wind direction result in non-elliptical fires. Head fire rate of spread is according to the fire behavior algorithms of McArthur (McArthur 1967; Noble and others 1980) with fuel loads modeled using Olson's (1963) model of biomass accumulation which has been parameterized for a range of Australian systems.

LAMOS(DS) is an implementation of LAMOS (Lavorel and others 2000) with a contagious spread fire model working on a daily time step. It is a simple model, sensible to daily minimum and maximum temperature, precipitation, fuel amount and slope. LAMOS(DS) contains two principle functions; one to estimate pan evaporation (Bristow and Campbell 1984; Roderick 1999) which, together with precipitation, produces a moisture budget, and a second equation to modify spread probabilities as a function of slope (Li 2000) and intensity. Fire intensity is the product of three linear functions: fuel load (0 – 1 kg m^{-2}), moisture (0-200mm) and temperature (5-25°C). Temperature during the course of the fire is interpolated between the daily minimum and maximum by a symmetrical sine function. Fires are assumed to begin when temperature is at the daily maximum. Fuel is consumed in proportion to the resulting intensity.

The LANDscape SUccession Model (LANDSUM) is a spatially explicit vegetation dynamics simulation program wherein succession is treated as a deterministic process, and disturbances are treated as stochastic processes (Keane and others 2002). Fire spread is a function of fuel-type, wind speed and direction, and slope using equations from Rothermel (1972) and Albini (1976). The elements that define the fire regime (for example average fire size, ignition probabilities) are input parameters, whereas fire regime is an emergent property for the other models. Ordinarily, the area burned in LANDSUM would not vary amongst the climate factors, however for this comparison, the probability of ignition success was made sensitive to the Keetch-Byram Drought Index.

The SEM-LAND model (Spatially Explicit Model for LANDscape Dynamics) simulates fire regimes and associated forest landscape dynamics resulting from long-term interactions among forest fire events, landscape structures, and weather conditions (Li 2000). A fire process is simulated in two stages: initiation and spread. The fire initiation stage continues from the presence of a fire ignition source in a forest stand until most trees in that stand have been burned. Once most trees are burned, the fire has the potential to spread to its surrounding cells. The probability of fire spread is determined by fuel and weather conditions and slope using relationships from the Canadian Forest Fire Weather Index system (Van Wagner 1987) and Canadian Forest Fire Behavior Prediction system (Forest Canada Fire Danger Group 1992; Hirsh 1996).

The Comparison Design

The comparison involved determining the sensitivity of modeled area burned to systematic variation in terrain, fuel pattern, climate and weather (Cary and others 2006). It incorporated three types of terrain, two types of fuel pattern, three different climates, and the full extent of weather variability for simulation locations. The simulation landscape was an array of 1000 by 1000 square pixels measuring 50 by 50 meters.

Variation in terrain was introduced by varying the minimum and maximum elevation of the simulation landscape by varying the amplitude of the two-dimensional sine function used to represent terrain. The sine functions had a periodicity of 16.67 km (333.3 pixels). Three landscapes representing flat, rolling and mountainous terrain, with maximum slope values of 0°, 15° and 30° respectively and relief of 0 m, 1250 m and 2500 m respectively were generated (figure 1). The average elevation of each landscape was 1250 m.

Fuel pattern was varied to represent finely clumped and coarsely clumped fuel patterns (figure 2). The finely clumped fuel pattern was comprised of ten by ten pixel (25 ha) clumps of varying fuel ages, whereas the coarsely clumped fuel pattern was comprised of fifty by fifty pixel (625 ha) clumps. Maps of fuel ages were generated by randomly allocating values from the series 0.1, 0.2, 0.3, ….1.0 to both finely and coarsely clumped fuel maps so that values were represented evenly across the landscapes. Ten replicate maps of each fuel pattern type were randomly generated for the model comparison. Fuel maps were transformed differently for each model to produce either fuel load or fuel age related maps that were meaningful to individual models (see Cary and others 2006). The maps of different fuel types were characterised by the same average fuel load or age, however the arrangement of different aged fuels varied between map types.

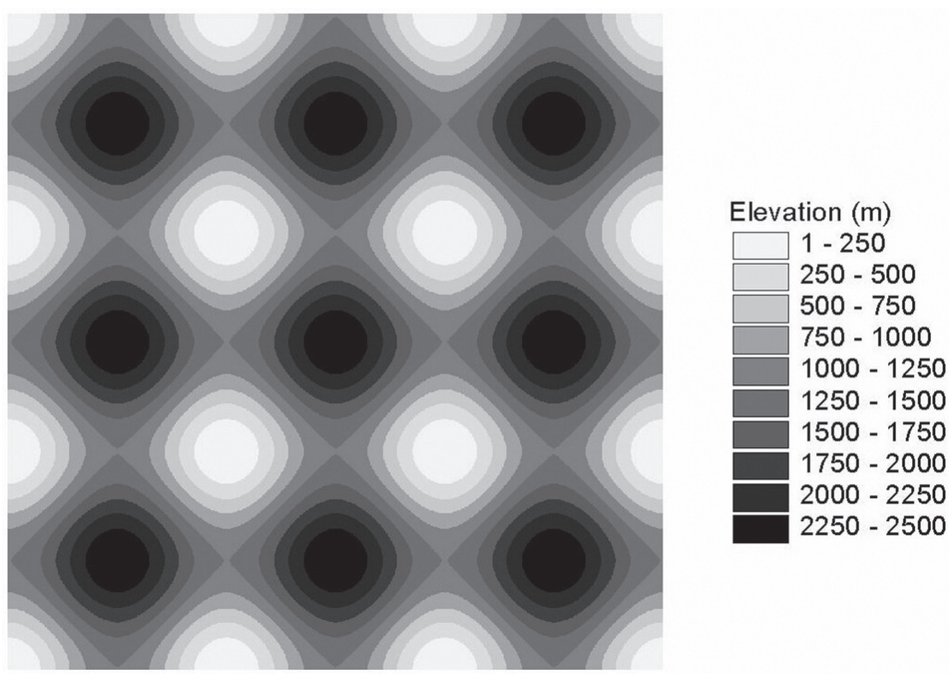

Figure 1—Pattern of elevation in mountainous landscape used in comparison of landscape-fire-succession models.

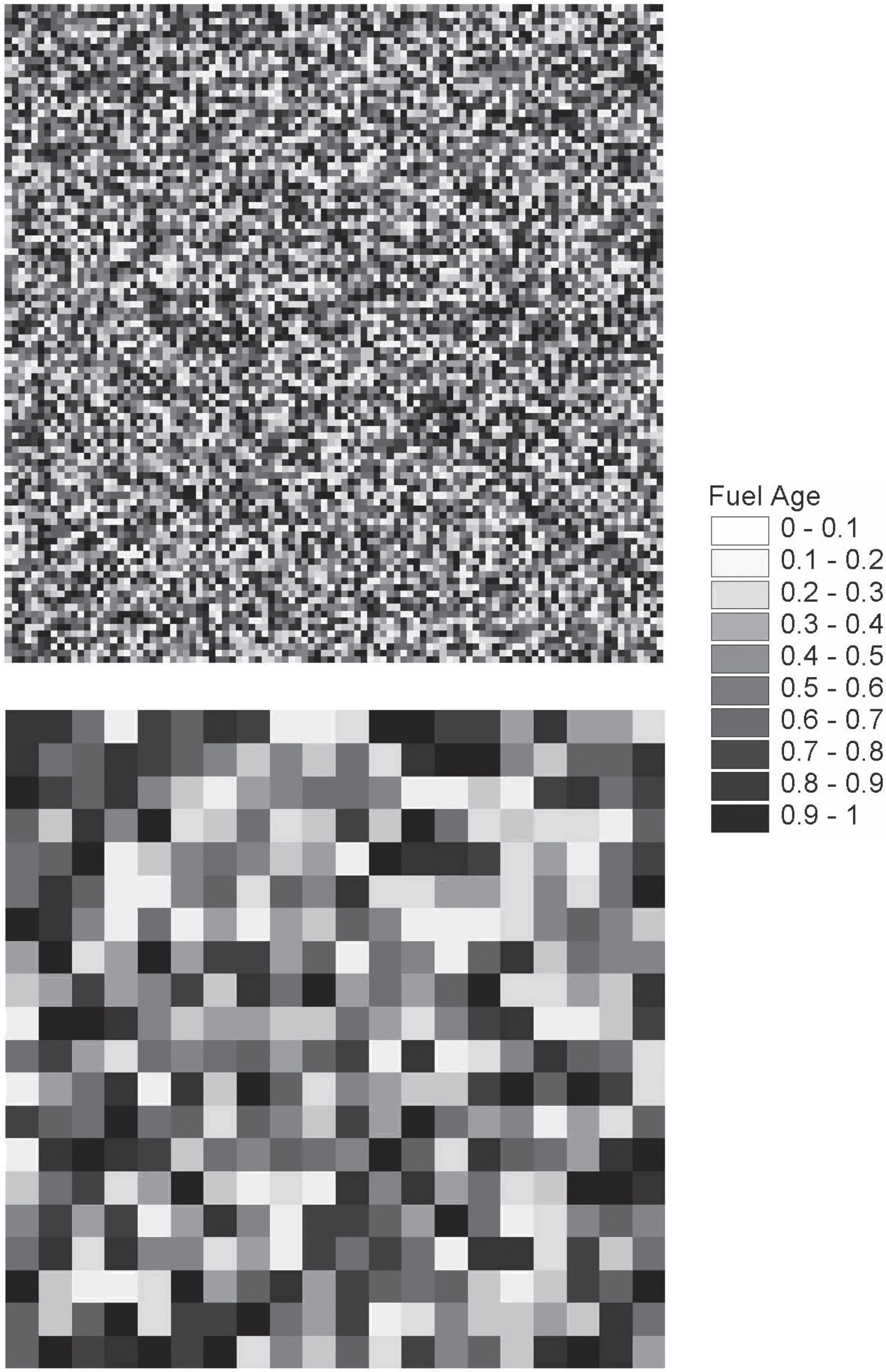

Figure 2—Replicate of each type of fuel pattern map used in comparison of landscape-fire succession models: a) finely clumped (25 hectare patches) and b) coarsely clumped (625 hectare patches) fuel pattern (values range from 0 to 1.0 and are transformed into fuel age or fuel load separately for each model.

Weather and climate are essentially different phenomena at fine temporal scales and were treated as orthogonal. Variation in weather was introduced for most models by selecting ten representative years of daily weather records for the landscape where the model has undergone most rigorous validation (table 1). For EMBYR, weather data from Glacier National Park, MT, was used. The ten weather years were selected so that the distribution of annual average daily temperature and annual average daily precipitation in the selected set best matched the variation in the weather record available (around 40 years for most models) (See Cary and others 2006). Three types of climate were included in the design, including observed, warmer/wetter, and warmer/drier climate. Daily values for the warmer/wetter and the warmer/drier climate were derived from the 10 weather years of observed climate by adding 3.6 °C (mid-range of projected global average temperature increase (1.4 to 5.8°C) (IPCC 2001) to daily temperature, and by decreasing daily precipitation by 20 percent for the warmer/drier climate and increasing daily precipitation by 20 percent for the warmer/wetter climate.

A total of 1,800 year-long simulations were run for each model (except for LANDSUM) from the 180 unique combinations of terrain (flat, mountainous, mountainous), fuel pattern (finely and coarsely clumped), climate (observed, warmer/wetter, warmer/drier), and weather (ten one-year replicates), given that there were ten replicate maps of each fuel pattern. Approximately 20 percent of the LANDSUM simulations did not experience fire and this resulted in a poor estimate of the probability and size of fires, because of the shortness of the simulation periods. This was rectified by performing ten

Table 1—Available weather data for study regions and associated models.

Location	Data type	Variables	Model
Glacier National Park, Montana	42 years, daily observations.	Daily maximum temperature (°C) Daily minimum temperature (°C) Daily precipitation (cm)	EMBYR LANDSUM
Edson, Alberta	34 years (1960 – 1993) of daily observation (observations at 1200 LST) from approximately the 1st April to 30th September, inclusive.	Temperature (°C) Relative Humidity (%) Windspeed (km.h^{-1}) Rainfall (mm) Daily FFMC*, DMC*, DC*, ISI*, BUI* Daily Fire Weather Index Number of days since rain * variables related to Fire Weather Index	SEM-LAND
Ginninderra, Australian Capital Territory	42 years of simulated weather based on Richardson-type weather simulator (Richardson, 1981) modified for all variables required for fire behaviour modelling.	Daily maximum temperature (°C) Daily minimum temperature (°C) Daily west-east wind speed (km.h^{-1}) Daily south-north wind speed (km.h^{-1}) Daily 9 am atmospheric vapour pressure (kPa) Daily precipitation (mm)	FIRESCAPE
Corsica	38 years (1960 – 1997) of daily observations.	Daily average temperature (°C) Daily precipitation (mm) Daily PET (mm)	LAMOS

simulation replicates for each unique combination of terrain, fuel pattern, fuel pattern replicate, climate, and weather replicate, and averaging them to produce a better estimate of area burned. Fires affected fuel load/age within each simulation but, since simulations were for only a single year, no vegetation succession algorithms were invoked. The total area burned per year (m^2) was recorded for each one-year simulation.

The sensitivity of simulated area burned to terrain, fuel pattern, climate and weather was assessed from the variance explained by each of the variables and all possible interactions. Variance explained (r^2) was determined from a fully factorial ANOVA performed in the SAS statistical package. Variance explained is a more meaningful measure than statistical significance when comparing the importance of environmental variables, particularly when dealing with simulated data. It facilitates the comparison of the importance of a range of variables on area burned, across a range of models with different input requirements and calibrated for widely separated landscapes characterised by quite different climate systems and weather syndromes. Plots of residual values against fitted values were constructed for each analysis. Analyses performed on untransformed area-burned data produced residuals which were highly skewed and the variance in residuals that was highly variable across fitted values. Transformation of area burned by the natural logarithm produced patterns of residuals that we considered acceptable for our analyses.

Results

Simulated area burned was more sensitive to climate and weather than to fuel pattern and terrain (table 2). Ln-transformed modeled area burned was considered sensitive to variation in climate for FIRESCAPE, LAMOS, LANDSUM and SEM-LAND while it was considered sensitive to variation in weather for EMBYR, FIRESCAPE, LANDSUM and SEM-LAND. The interaction between these two variables was considered important for EMBYR, LANDSUM and SEM-LAND. For models sensitive to climate, there was a trend for increasing area burned for warmer climates (warmer/drier and warmer/wetter) compared with the observed climate, with the warmer/drier climate being characterised by larger area burned than the warmer/wetter climate in two of four cases (see Cary and others 2006).

Only FIRESCAPE showed sensitivity to variation in terrain (and the interaction between terrain and weather, and that between terrain, climate and weather). Modeled area burned was highest for mountainous terrain and least for flat terrain. Only EMBYR showed sensitivity to variation in fuel pattern (and the interaction between fuel pattern and weather factors). Modeled area burned was higher for the coarsely clumped fuel pattern than for the finely clumped pattern (see Cary and others 2006).

Discussion

The variance in modeled area burned was greater for weather than climate for EMBYR, LANDSUM and SEM-LAND, compared with FIRESCAPE and LAMOS, perhaps because the inter-annual variation between the weather years for these locations was lower than for other sites. Nevertheless, sensitivity of modeled area burned to weather was considered important for four

Table 2—Relative Sums of Squares attributed to different sources of variation in the comparison of sensitivity of ln-transformed area burnt to terrain (Terrain), fuel pattern (Fuel), climate (Climate) and weather factors (Weather), and their interactions. Factors and their interactions are considered important if they explain more than 0.05 and 0.025 of total variance respectively. Factors and interactions considered unimportant are blank. Significant factors and interactions (P < 0.05) are indicated by *.

		Model				
Source	**DF**	**EMBYR**	**FIRESCAPE**	**LAMOS**	**LANDSUM**	**SEM-LAND**
Terrain	2		0.293*			
Fuel	1	0.217*	*		*	*
Terrain x Fuel	2		*			
Climate	2	*	0.418*	0.278*	0.178*	0.370*
Terrain x Climate	4		*			
Fuel x Climate	2	*				*
Terrain x Fuel x Climate	4		*			
Weather	9	0.329*	0.087*	*	0.333*	0.542*
Terrain x Weather	18		0.025*		*	
Fuel x Weather	9	0.031*	*			*
Terrain x Fuel x Weather	18	*				
Climate x Weather	18	0.096*	*	*	0.224*	0.046*
Terrain x Climate x Weath	36		0.025*			
Fuel x Climate x Weather	18	*				
Terr x Fuel x Clim x Weath	36					
Model	179	0.744	0.905	0.401	0.766	0.971

Note that not all significant sources are considered important.
(Source: Cary and others 2006)

out of five models. The overriding importance of weather for fire activity has been highlighted in numerous studies (see Flannigan and Harrington 1988; Swetnam 1993; Bessie and Johnson 1995; Hely and others 2001; Flannigan and Wotton 2001). Our finding regarding the importance of weather across a range of models highlights the importance of adequately incorporating variability in weather into landscape-fire-succession models.

Several authors have provided simulated evidence for increasing area burned or frequency of fire under warmer climates (Clark 1990; Cary and Banks 1999; Li and others 2000; Cary 2002), possibly due to a longer fire season (Stocks et al 1998; Wotton and Flannigan 1993). This is consistent with our general findings. Climate was not considered important for EMBYR although earlier studies have indicated that a wetter climate would result in larger fires (Gardner and others 1996). A possible explanation for the discrepancy is that, in this study, simulations were only one year in length and vegetation succession effects were not incorporated. We are planning new research where simulations will be centuries long, allowing for the importance of vegetation succession to be explored.

Fuel pattern was relatively unimportant, except in the case of EMBYR. Fire spread in EMBYR is partly a function of the nature of fuel in the source and target pixels of any fire spread event. Frequently changing fuel condition in the finely clumped fuel pattern resulted in a decrease in area burned compared with the coarsely clumped pattern. While this is a realistic representation of fire spread, fuel pattern accounts for a comparatively small amount of variance in EMBYR compared to climate and weather in the other models.

Terrain was considered important for FIRESCAPE, despite all models incorporating a similar positive effect of slope on fire spread. FIRESCAPE is the only model that varies weather with terrain. The mountainous terrain provides a greater proportion of the landscape which is warmer and drier (in the "valleys"), compared to the rolling and flat landscapes, given that all landscapes were characterized by an average elevation of 1250 m. Representing the effect of terrain on weather in landscape fire models is fundamental if this aspect of the terrain factor is to influence models results in a realistic fashion.

Our results have implications for representing fire in higher-order models like Dynamic Global Vegetation Models (DGVMs). The relative unimportance of fine scale fuel pattern indicates that coarse scale DGVMs may not need to incorporate pattern of vegetation within simulation cells, although this depends on the importance of vegetation succession on area burned, which was not tested in this experiment. On the other hand, landscape scale pattern in terrain was demonstrated to be fundamentally important using the one landscape-fire-succession model that incorporates the effect of terrain on weather. Also, the general finding of the importance of inter-annual variability in weather (compared with climate) has important implications for the inclusion of fire into DGVMs because an increase in inter-annual weather variability resulted in greater effects on area burned than the climate variable in some cases.

The results from this study are concerned with comparing landscapes where the mean fuel age/load is constant across simulations but varies in the arrangement of fuel (fuel pattern). We are presently using our approach to compare the sensitivity of modeled area burned to variation in approach/extent of fuel management and ignition probability. It also has considerable potential for conducting comparisons amongst groups of other types of models producing variation in landscape dynamics, and for further comparison amongst landscape-fire succession-models.

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Assessing Ecological Departure from Reference Conditions with the Fire Regime Condition Class (FRCC) Mapping Tool

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***Abstract**—Knowledge of ecological departure from a range of reference conditions provides a critical context for managing sustainable ecosystems. Fire Regime Condition Class (FRCC) is a qualitative measure characterizing possible departure from historical fire regimes. The FRCC Mapping Tool was developed as an ArcMap extension utilizing the protocol identified by the Interagency Fire Regime Condition Class Handbook to derive spatial depictions of vegetation departure. The FRCC Mapping Tool requires a biophysical setting layer identifying potential vegetation distribution, a current succession class layer allowing for comparison with historical vegetation, and a landscape layer (assessment area boundaries) as input data. The tool then compares existing vegetation composition for each biophysical setting to previously modeled reference conditions for those types. As described in this paper, spatial outputs characterizing vegetation departure at the succession class, biophysical setting, and landscape levels can be used by land managers to identify restoration objectives and priorities.*

Introduction

Severe wildfires in recent years have prompted Federal action to protect communities and restore landscapes and associated fire regimes (USDA Forest Service 2000). A standardized, relatively simple method of landscape assessment was needed to measure progress in ecosystem restoration (Schmidt et al. 2002). The Fire Regime Condition Class (FRCC) assessment method was developed (Hann et al. 2005) to meet this need, and to evaluate departure from a range of reference conditions at multiple scales. Reference conditions include the median values for abundance of seral stages, as well as an estimate of historical fire frequency and severity on landscapes and are developed for each BpS. FRCC is a classification of the amount of departure of conditions at a given time period (such as current or future) from historical ecological reference conditions (Hann et al. 2005). Current policy direction for federal lands management, embodied in the Healthy Forest Restoration Act of 2003 (P.L. 108-148), requires FRCC assessments as part of pre-restoration planning and post-restoration monitoring.

Because of the prominence of FRCC in legal and administrative direction, a number of national and regional trainings in FRCC methods were conducted in 2003 and 2004, with the aim of improving understanding and implementation of FRCC assessments. FRCC training continues at the local level, and is also available on line at www.frcc.gov. An understanding of these methods is a necessary precursor for effective use of the FRCC Mapping Tool.

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Central to the FRCC concept is a classification of landscape integrity relative natural or "reference conditions." We define natural conditions as the range of ecological structure, function, and composition operating on landscapes without post-European settlement influence. Because of uncertainties and lack of information on what this range would be at present, we use the historical range of variation (that prior to European settlement) as an approximation of what the current natural range would be. Given the constraints of currently available data and knowledge, this historical range of variation (HRV) is assumed to represent the best understanding of a properly functioning ecosystem (Landres et al. 1999, Hessburg et al. 1999). When actual historical data are available (tree ring studies, legacy photographs, etc.), the historical range of variation can be described directly, if often incompletely. Usually, however, modeling is required. Modeling this range of historic reference conditions, and then comparing it to current conditions, allows us to infer a departure from conditions presumably influenced by a properly functioning disturbance regime (Cleland et al. 2004).

Moving landscapes closer to the historic range of variation can be useful if the management goal is to restore ecosystems across landscapes. Note, however, that the range of variation is not necessarily the same as a desired future condition. Maintaining wildlife habitat and protecting communities from wildlfire risk are examples where management goals are not necessarily the same as moving landscapes towards HRV.

A simple, intuitive concept in principle, modeling HRV can be fraught with complexity and sources of error. One problem with estimating historic landscapes is that we are generally working with very little data (Gill and McCarthy 1998, Dillon et al. 2005, Marcot 2005). Another problem is that climate change may lead to changing reference conditions; i.e., the historical range of variation becomes obsolete as an approximation of the natural range of variation. Nevertheless, HRV remains our best approximation of a properly functioning system, at least until better models are available.

Dillon et al. (2005) cautioned that modeling HRV has four primary requirements: 1) analyses should be conducted at multiple scales so that important ecological processes are not missed or misrepresented; 2) assessments should consider spatial variation of vegetation patterns across landscapes (see also Arno and Petersen 1983, Johnson and Gutsell 1994); 3) variability can be calculated in several ways, and this should be considered for a more meaningful result (see also Marcot 2005); and 4) consider the role of climate change over time; e.g., climatic conditions during the Little Ice Age (1700-1850), a timeframe often used for the historic range, are very different from those today (see also Millar and Woolfenden 1999).

The FRCC Mapping Tool is a menu-driven GIS extension automating and spatially applying FRCC calculations. As designed and with subsequent refinements, it addresses each of these considerations. The practical outcomes of Mapping Tool use, however, are still unfolding as it is implemented and results evaluated. The Mapping Tool can be easily run at multiple scales, providing that input layers are delineated or can be aggregated at those scales, addressing requirement (1) above. FRCC is based largely on variation in spatial patterns, addressing requirement (2). Throughout this paper, the reader should fully realize departure is calculated using an estimated mean or median value of succession stage abundances. Departure from a range of values would be more meaningful, and methods to develop this are under active consideration (requirement 3). Finally, as for climate change (requirement 4), there is nothing in FRCC that precludes modeling different climate scenarios. As climate change effects on vegetation become better understood and models more widely available, FRCC reference conditions can be adjusted accordingly.

During the initial development of the FRCC methodology, and with subsequent research efforts such as the multi-year LANDFIRE project (www.landfire.gov), reference conditions were modeled to estimate HRV. Specifically, HRV was estimated for vegetative structure and composition, and in terms of fire regime characteristics (fire frequency and severity). Using a combination of literature searches, expert opinion, and simulation modeling, HRV metrics were developed for all major vegetation types, or "Biophysical Settings" (BpS), in the U.S. Biophysical settings are a potential vegetation concept defined using a disturbance-constrained approach; i.e., succession and vegetation development occur within the bounds of historic natural disturbances; non-lethal disturbance frequency and severity can influence successional trajectories (Hann et al. 2005). To date, more than 300 reference condition models provide the basic foundation for diagnosing FRCC at multiple spatial scales.

The FRCC system is an index of departure, with three condition classes. Properly functioning landscapes, defined as exhibiting less than 33 percent departure from the median or average HRV conditions, receive a Condition Class 1 rating. Condition Class 2 represents landscapes with moderate departure (33 to 66 percent departure), and Condition Class 3 lands show high departure (greater than 66 percent). These classes are generally useful for planning and prioritizing ecosystem maintenance and restoration. For example, FRCC data might provide baseline data for pre- and post-treatment planning, monitoring, and accomplishment reporting.

FRCC assessments can be conducted in several ways. Field-based assessments can be made where an evaluator rates the vegetation (succession stage abundance) and fire regime components (current fire frequency and severity) of the landscape using aerial photography, field observation, and fire atlas data. These landscapes are generally in the range of hundreds to thousands of acres. This method is useful for field checking of estimates made at broader scales and for local monitoring. Another alternative is to use the FRCC Mapping Tool with remotely sensed vegetation data in a geographic information system (GIS) to produce maps at various scales. The Mapping Tool evaluates remotely sensed vegetation data to produce spatially specific FRCC diagnoses. A third option, not discussed in this paper, is to download the remotely sensed FRCC map from www.landfire.gov. That data layer, however, was designed for regional and national-scale analyses and may be too coarse for many analyses.

The FRCC Mapping Tool provides an objective, consistent, and spatially specific way to measure post-European settlement changes across multiple geographic scales if suitable data are available. Assessments based on the FRCC Mapping Tool can help managers prioritize landscapes for possible restoration and maintenance activities from fine (e.g., hundreds of acres) to coarse (e.g., millions of acres) scales. Finally, the Mapping Tool is relatively easy to use and understand—not a minor consideration when a standardized method for use at multiple organizational levels is needed.

FRCC Mapping Tool Characteristics

The FRCC Mapping Tool was designed in conjunction with the field-based Standard Landscape Method described in the FRCC Guidebook (Hann and others 2005). In contrast with field-based FRCC assessments, the Mapping Tool is a GIS application that produces multiple spatial layers to analyze pixel- to landscape scale (ranging from hundreds to millions of acres) departure and FRCC.

Both FRCC methods use similar principles to evaluate landscape departure and condition class. Field-based assessments evaluate existing vegetation and fire frequency/severity, whereas the FRCC Mapping Tool currently assesses only the departure of existing vegetation from reference vegetation conditions. To date, the software team developing the mapping tool has not been able to develop a way to effectively evaluate post-European settlement fire frequency and severity for a given landscape. This is primarily because these data layers are lacking or inconsistent for most areas of the country, not because of software limitations. Nonetheless, for many biophysical settings the existing condition indicates changes in fire regimes compared to the reference range.

Because of the similarity between the two FRCC methods, potential users of the Mapping Tool should first seek FRCC certification (see www.frcc.gov). In addition, users should have a firm understanding of geographic information systems (GIS) and experience using raster data and ArcMap (Version 9.0 or later) software. The Mapping Tool software, user guide, and systems requirements can be downloaded at www.frcc.gov.

The FRCC Mapping Tool uses three input layers to produce six output layers. (See Figure 1 for a diagram of the mapping process used in the Tool.) The Mapping Tool also produces a summary spreadsheet known as the Management Report. This report shows the current acres in each BpS succession class, and the area that would need to be converted to restore a landscape with a range of conditions similar to the historical range.

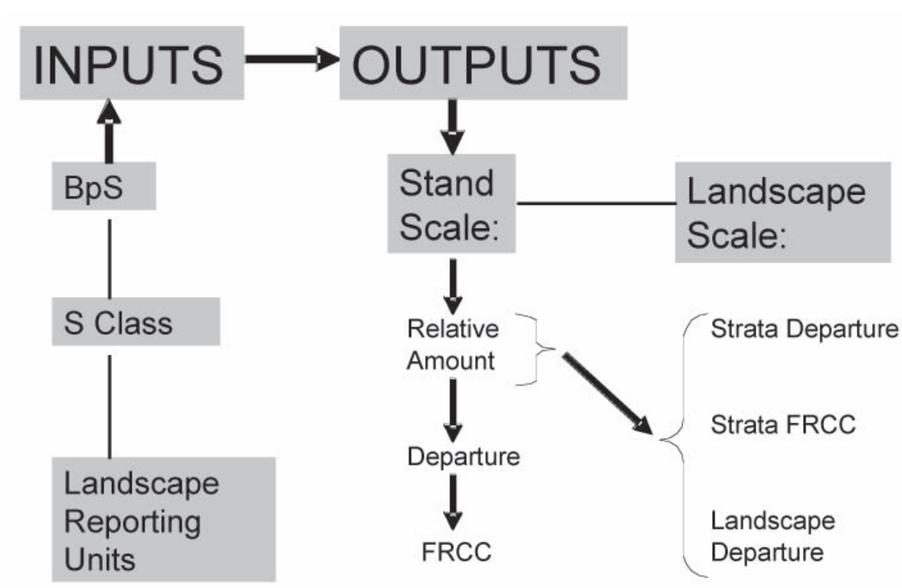

Figure 1—Diagram of the FRCC Mapping Tool process.

Input Data Layers

The Mapping Tool derives its suite of FRCC attributes from three user-provided input layers. These data sources can range widely, from coarse field-level data, to data derived from satellite imagery, to photo-interpreted vegetation mapping with extensive field checking. Because FRCC is a scale-dependent variable (Hann and others 2005), users must first provide a map to support scale-appropriate succession class analysis. This *Landscape Layer* should identify the appropriate spatial scale and boundaries for assessing FRCC. It may vary by BpS or geographic area. The Mapping Tool allows up to three landscape levels for consideration. For example, a tri-level nested hierarchy of area hydrologic units or similar nested classification can be used. When based on hydrologic units, for example, the map units might range from subwatersheds, to watersheds, to subbasins (nested watersheds of increasing area, Figure 2). These hierarchical maps allow the FRCC Mapping Tool to analyze Succession Classes according to ecologically appropriate scales, which differ among fire regimes. For example, a subwatershed scale can be used where small or patchy fires predominated historically (fire regime groups I and II [Hann and others 2005]). Conversely, BpS's influenced primarily by large replacement fires (Regimes IV and V) should be analyzed at the largest landscape scale because large fires can falsely appear to skew the statistical distribution of succession classes for small study areas. Hann and others (2005) have developed guidelines for analyzing FRCC based on fire regime-topography combinations (Table 1).

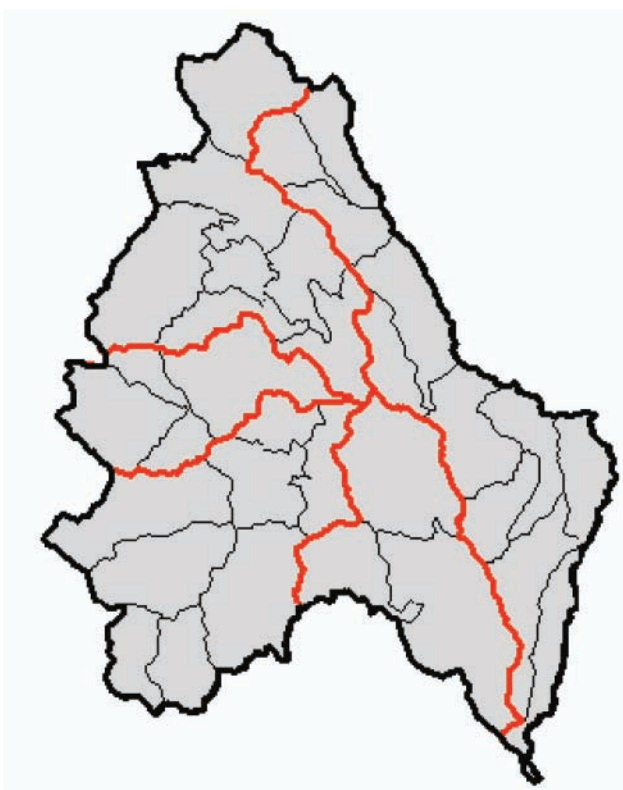

Figure 2—Example of tri-nested landscape hierarchy based on hydrologic units (from Hann et al. 2005). Such ecologically based classifications are useful for FRCC analysis, where potential analysis units range from the subwatershed to the subbasin scales.

Table 1—Scale guidelines for determining FRCC (Hann and others 2005). Suggested analysis area size range is based on dominant fire regime type and is inversely related to slope steepness and land dissection.

	Terrain	
Fire regime group[1]	**Flat to rolling (lightly to moderately dissected)**	**Steep (moderately to highly dissected)**
	acres	
I, II	50-2000	50-1000
III	500-2000	250-1000
IV,	5000-1,000,000	2000-250,000
V (replacement severity)	5000-1,000,000	2000-250,000
V (mixed severity)	50-10,000	50-10,000

[1] I (0-35 yr/low to mixed severity); II (0-35 yr/stand replacement); III (35-200 yr/mixed severity); IV (35-200 yr/stand replacement); V (200+ yr/stand replacement [but can include any severity type]).

To summarize input requirements for the landscape layer, the user must: 1) provide a base map containing up to three nested landscape sizes, such as hydrologic units or ecological units (Winthers et al. 2005), and 2) in an associated table, specify for the Mapping Tool which landscape levels are appropriate for FRCC analysis based on BpS, dominant fire regime types and associated terrain dissection. The Mapping Tool then concurrently analyzes BpS vegetation succession classes according to each user-specified landscape level in the area.

The FRCC Mapping Tool also requires a *Biophysical Settings* input layer, which shows BpS distribution within the analysis area. The Mapping Tool analyzes this layer in tandem with a user-provided Reference Condition table to document the estimated average amount of each succession class historically. For instance, results from a given BpS model might suggest up to 20 percent of the type occurred in the early seral succession class, 40 percent occurred in the mid-seral open class, 10 percent occurred in the mid-seral closed class, and so on.

The LANDFIRE reference condition tables for the entire U.S. will load automatically after installing the Mapping Tool software, or users can develop custom reference condition tables based on local data. These tables must contain three pieces of information for the Mapping Tool: 1) a comprehensive list of all BpS within the study area, 2) reference condition amount (in percent) for each BpS succession class, and 3) the appropriate landscape reporting scale for each BpS type. Determining this scale generally means identifying a scale large enough to encompass the normal range of disturbance (fire) sizes and frequency for the question of interest.

Finally, the user must provide a *Succession Classes* layer showing the current distribution of succession classes within the analysis area. This layer can be generated from local current vegetation layers crosswalked to the appropriate FRCC succession class. This allows the Mapping Tool to compare the current amount of each succession class to the estimated historical amounts, thus assessing FRCC departure and condition class diagnoses. The LANDFIRE project represents a good source of data for succession class and other information. Upon completion in 2009, comprehensive U.S. map coverage will be available for succession classes, BpS, and other layers.

Output Data

To date, the FRCC Mapping Tool produces six output raster (pixel-based) GIS coverages (map layers) describing various Fire Regime Condition Class metrics. The Mapping Tool also generates a report summarizing the raster data. Two additional rasters are now in the final stages of development, as discussed below. For more detailed information on all layers, see the FRCC Guidebook (Chapter 4 in Hann and others 2005).

Output layers generated by the Mapping Tool fall into two groups: those at the BpS/landscape scales and those at the succession class/stand scales. The first group (BpS/landscape scales) includes three layers. The first of these, the *Strata Departure* layer summarizes Departure for each BpS, (or landscape "stratum," Hann et al. 2005). (Note that the soon-to-be-replaced FRCC Guidebook uses the now outdated name "S*tratum S-Class Departure*" for this layer.) The *Strata Departure* layer integrates the landscape strata according to a number of percent Departure classes. The next layer is the "*Strata FRCC*" layer (previously called the "*Stratum S-Class FRCC*" layer) (Figure 3). This data layer classifies the various BpS departure results according to the three FRCC Condition Classes described above. The final raster currently available is the "*Landscape Departure*" layer. Here, the Mapping Tool rates landscape-scale Departure by calculating an area-weighted average of the various strata departure percents, then by generating an overall rating for the appropriate landscape scale. When an area is dominated by large replacement fires, for instance, the tool bases the departure rating on the largest landscape scale defined by the user, such as a watershed occupying tens of thousands of acres.

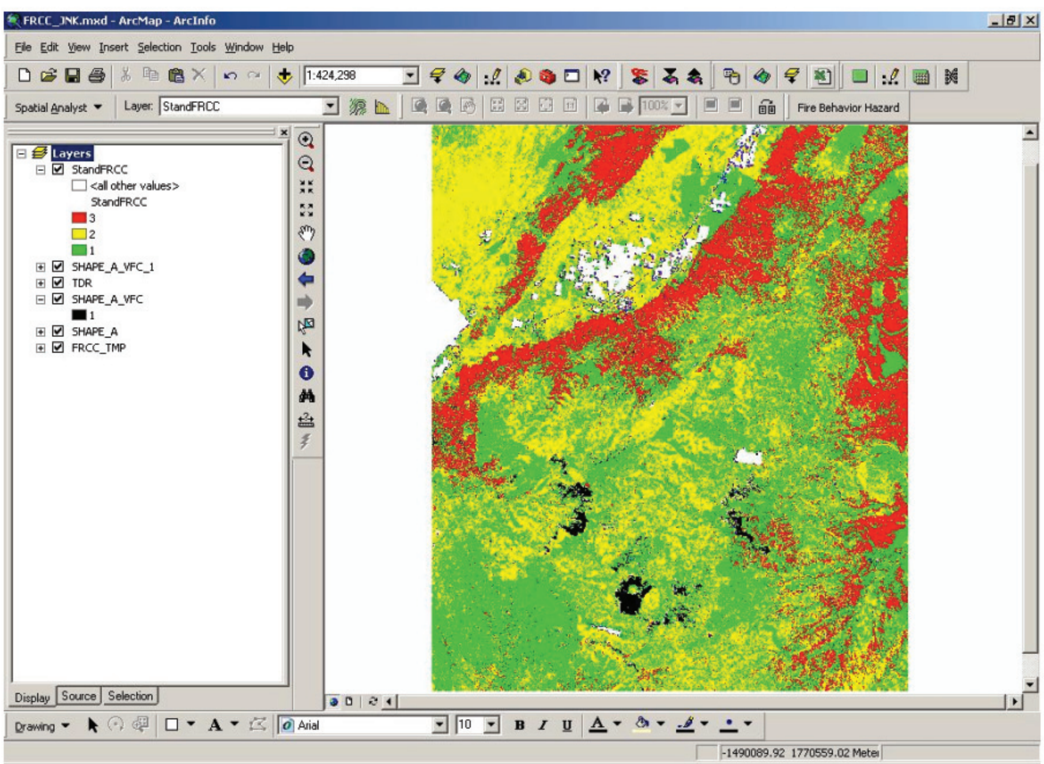

Figure 3—Example of FRCC Mapping Tool output for a hypothetical analysis area. Map shows Fire Regime Condition Class for the various landscape Strata, which typically represent an area's biophysical settings (*Key*: green is Condition Class 1, yellow is Condition Class 2, red is Condition Class 3 [white polygons indicate "No Data"]).

In the second group (succession class/stand scales), the first data layer generated by the FRCC Mapping Tool is the *Succession Class Percent Difference* layer. This output compares the amount of each BpS succession class during the current period to the estimated average amounts for the Reference period. In this case the measurement scale ranges from -100% to +100%, with zero representing similar amounts, negative values indicating deficient amounts, and positive percents representing excessive amounts. That is, the layer shows the most deficient to the most excessive (relative to the historic median) succession classes on today's landscape.

The next output layer is the *Succession Class Relative Amount*. (The current version of the FRCC Guidebook (Hann et al. 2005) uses the now outdated name "Stratum S-Class Relative Amount" for this layer.) This layer simply classifies the percent difference data according to the FRCC Guidebook (Hann and others 2005)(Figure 4). For example, pixels with a percent difference value of between minus 33 and minus 66 percent are "under-represented," whereas values between plus 33 and plus 66 percent are considered "over-represented." Classifying the myriad results from the percent difference layer thus helps users more easily identify which succession classes should be maintained, versus those that could be reduced or recruited, in order to emulate average BpS Reference Conditions.

Finally, the *Stand Condition Class* (FRCC) layer, previously called "*Stand Level FRCC*" (Hann et al. 2005), further classifies the above results. Here, the Mapping Tool rates the relative amount output according to the three Condition Classes mentioned earlier. For example, pixels in the "similar," "under-represented," and "trace" relative amount classes are rated as Stand Condition Class 1. Pixels in the "over-represented" relative amount class are considered to be Stand Condition Class 2, and those in the "abundant" relative amount class are Stand Condition Class 3. This layer was developed primarily to facilitate reporting and accomplishment. We stress this layer should not be used as a proxy for the landscape condition class layer, because the latter is a more appropriate layer for identifying FRCC, a landscape-scale measure. It is better to think of stands as having *membership* in successional stage classes that are either over-abundant, under-abundant, or within the historic range.

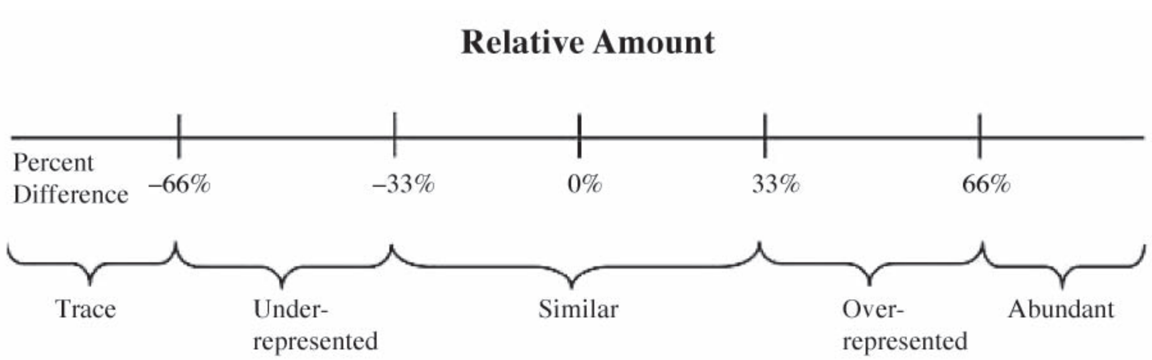

Figure 4—The Percent Difference- and Relative Amount scales used for FRCC assessments.

Software for two additional rasters currently is being developed, yielding an eventual total of eight data layers. Specifically, a *Stand Departure* layer and a *Landscape Condition Class (FRCC)* layer will likely be available by late 2006. The *Stand Departure* layer will base departure at the local (stand) scale on each stands membership in an seral stage abundance class compared to the historic average. The *Landscape Condition Class* layer will generate a single FRCC call for a landscape (delineated by the user) that is the weighted average of its member *Strata Condition Classes.*

The FRCC Mapping Tool also generates a Management Report spreadsheet to accompany the output rasters. The spreadsheet serves as the primary tool for analyzing and interpreting the GIS results, helping to support various planning needs. For instance, the data helps identify the ecological condition of an individual BpS or for multiple BpS in a given analysis area. The GIS data can also help managers identify ecological conditions and prioritize treatments ranging in scale from individual stands to entire landscapes. Such FRCC data can also be useful for fulfilling various reporting requirements, for developing budgets, and for supporting public education.

Mapping Tool Limitations

The FRCC Mapping Tool has several limitations. First, unlike field-based assessments, the Mapping Tool cannot be used to document post-settlement trends in fire frequency and severity. In many cases, however, the remotely sensed vegetation condition serves as an indirect measure of current fire potential, essentially serving as a proxy for those two FRCC metrics. Using remotely sensed data to identify numerous vegetation types and current conditions also can be difficult. Distinguishing between closely related BpS types and among the various succession classes is frequently challenging, particularly when types occupy closely similar terrain. In the western U.S., for example, the distinction between early successional Class "A" in pinyon pine (*Pinus edulis*)-juniper (*Juniperus* spp.) woodlands and similarly grass-dominated succession classes in adjacent sagebrush (*Artemisia* spp.) types can be difficult, especially for broad ecotones. Identifying various types of FRCC-defined "Uncharacteristic" succession classes also can be difficult when using remotely sensed data. Examples include areas invaded by varying amounts of exotic cheatgrass (*Bromus tectorum*), and woodland-grassland ecotones experiencing tree encroachment as a result of post-1900 fire exclusion. To help mitigate such interpretation errors, users of the FRCC Mapping Tool might need to conduct local field sampling to help improve the digital "signatures" for the remotely sensed data.

Management Applications

To date, land managers have used the FRCC Mapping Tool to support various planning activities. Introduced in late 2004 during a number of training sessions in the western U.S., the FRCC Mapping Tool is gaining acceptance and use. Although the Tool has not yet been fully implemented, enough practical experience has emerged that we can highlight several management oriented examples and issues here. As of 2006, the mapping tool has been used to determine FRCC on National Forests throughout much of the Pacific Northwest Region. One of the software's main strengths as reported by users is the personnel time saved with its use. The Tool has helped automate a GIS process that would otherwise require a number of time-consuming steps. The FRCC Mapping Tool has also helped promote a standardized approach to determining FRCC (Jane Kertis, Siuslaw National Forest, pers. comm.), facilitating communication among land managers.

Improper or inconsistent use of the Mapping Tool, rather than software design and function, seems to be the main issue to date. The Mapping Tool will not run if the input layers do not agree with each other and with the reference condition table. For example, if a BpS on the map layer is not included in the reference conditions table, the software will not run. Hence the importance of consistent input data without errors. Also, using inappropriate landscape input maps can be expected to produce varying degrees of FRCC estimation error for similar vegetation types. Experienced users are currently helping to educate their peers about the FRCC scale issue and the appropriate uses of the Mapping Tool. Instructions on use of the Mapping Tool can be found in the FRCC Guidebook (Hann et al. 2005).

The FRCC Mapping Tool will be used to assess subregions, such as northwest Oregon (Jane Kertis, Siuslaw National Forest, pers. comm.). Similarly, the USDA Forest Service Pacific Northwest Region's standardized existing vegetation mapping effort, known as the Interagency Mapping and Assessment Process (IMAP) also will examine the potential utility of the Mapping Tool for assessing FRCC and related metrics at more local landscape scales than LANDFIRE does. Given the vast amount of area in the U.S. currently in need of ecological assessments, newly emerging GIS software such as the FRCC Mapping Tool will become increasingly important to land managers.

Acknowledgments

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Predicting Post-Fire Severity Effects in Coast Redwood Forests Using FARSITE

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Abstract—*Assessing post-fire impacts in coast redwood (Sequoia sempervirens) forests can be difficult due to rough terrain, limited roads, and dense canopies. Remote sensing techniques can identify overstory damage, locating high intensity damage areas, although this can underestimate the effects on the understory vegetation and soils. To accurately assess understory impacts requires field assessment techniques, which can be expensive for larger burn areas. Where geospatial data for fuels and topography can be combined with weather data using FARSITE, a fire behavior simulation model, landscape fire behavior predictions can be made. Fire behavior outputs can be generated to produce a post-fire predicted landscape map of fire severity. The 2003 Canoe fire burned 4,000 hectares, primarily in old-growth redwood forests in Humboldt County, California. Post-fire sampling of burn impact was assessed using the Composite Burn Index methodology and found to be unrelated to FARSITE produced fire behavior variables using regression analysis. This finding is understandable because basic FARSITE landscape data available for this fire lacked fuel load information for post-combustion analysis. The Canoe Fire had a slow rate of spread, and with the deep fuel beds present; long duration burning was observed. Fire severity, as described by the Composite Burn Index, was greatest in the forest understory. FARSITE was a useful projection tool for perimeter advance and flame lengths associated with the fire front.*

Introduction

The short-term effects of wildfire on vegetation, soils, wildlife, and watersheds are poorly understood in the coastal redwood [*Sequoia sempervirens* (D. Don) Endl.] forests of northern California. The September 2003 4,575 hectare, (11,214 acre) Canoe Fire, ignited by lightning in Humboldt Redwoods State Park, provided a rare opportunity to better understand the mixed effects of fire following logging and over a half century of fire exclusion in old-growth and second-growth forests.

Assessing post-fire impacts in coast redwood forests can be difficult due to rough terrain, limited access, and dense canopies. Remote sensing techniques can identify overstory damage, locating high intensity damage areas, although this can underestimate the effects on the understory vegetation and soils. To accurately assess understory impacts requires field assessment techniques, which can be expensive for larger burn areas.

Where geospatial data for fuels and topography can be combined with weather data using a fire behavior simulation model, landscape fire behavior predictions can be made. Fire behavior outputs can be generated to produce a post-fire predicted landscape map of fire severity.

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Methods

The 2003 Canoe fire started in Humboldt Redwoods State Park in Humboldt County, California, burning primarily old-growth and young-growth redwood forests. Stand species included coast redwood, Douglas-fir (*Psudeotsuga menziesii)*, and tanoak (*Lithocarpus densiflorus*) in the overstory. Understory species included suppressed redwood, tanoak, huckleberry (*Vaccinium* sp.) and *Oxalis oregana*. The burn included unlogged old-growth areas, partially logged areas with a residual old-growth component, and previously logged areas that have stands of 60 to 100 year young-growth. A field based fire severity assessment was completed 9 months after the burn using the Composite Burn Index Methodology (FIREMON 2003) and was used to calibrate a map of the fire effects based on remotely sensed data. We tested the prediction ability of the FARSITE (Finney 2004) fire simulator to produce a similar map.

CBI Analysis

An initial fire severity map was created using a remote sensing approach. Pre- and post-fire IKONOS imagery (2002 pre-burn versus 2004 post-burn) was visually compared to delineate fire severity boundaries. Oblique imagery taken after the fire from a helicopter in December 2003 was used to validate three established severity classes. Severity classes were defined as: low with no visible change to the canopy; medium with <50% canopy loss; and high with a >50% loss. The minimum mapping unit was approximately 5 acres and boundaries were drawn with heads-up digitizing.

The forests of the burned area were classified into one of three community types (alluvial redwood, slope redwood and Douglas-fir forests), two management histories (old-growth or second growth stands), and two fire severity types based on observations of canopy conditions, with low representing green canopy conditions, and high with canopy mortality. This design created a factorial of 12 stand types and five replicate plots that were assigned at random using a GIS application. One type, old-growth alluvial high severity did not exist and therefore was excluded. As a result, 55 plots were installed and utilized for comparisons.

The Composite Burn Index (CBI) is a field technique developed by the interagency FIREMON program to identify and quantify fire effects over large areas. FIREMON is designed for repetitive measures. We applied the CBImethodology during the summer of 2004, nine months post-burn in 0.04 hectare (0.1 acre) circular plots. Characteristics were related to individual strata and scores averaged for the whole plot. The strata consisted of a) substrates or soils, b) herbs, low shrubs and small trees < 1 meter tall, c) tall shrubs and saplings 1< 5 m, d) intermediate and subdominants trees, and e) the dominant trees. The color and condition of the soils, the amount and quality of the fuels and vegetation consumed, the regeneration post-fire, the establishment of new seral species, and blacking, scorching and torching of the trees was evaluated.

We used the field data to calibrate or validate the remote sensing results. Our results are presented as an average of the scores for 1) total plot (i.e. all strata) 2) overstory (i.e. only the dominant tree stratum) and 3) understory (i.e. soil to vegetation <5 m tall). The CBI produces a score on a 0-3 basis with 3 as extremely high severity.

NCSS was used to analyze the data using ANOVA and means separation was performed with Fisher's Protected LSD.

FARSITE Analysis

FARSITE is a spatial fire behavior simulation system. The base landscape data was created at the Northern California Geographic Area Command Center, Redding, California in September 2003, and was used during the fire to predict short and medium range fire growth. Slope, aspect, and elevation data are derived from 30-meter resolution USGS Digital Elevation Models. The fuel model layer was derived from the California Department of Forestry and Fire Protection's Forest and Rangeland Assessment Program remote sensing data. Crown canopy values were estimated by H. Scanlon during the fire. No fuel loading data for post-frontal combustion analysis was available.

Weather data for analysis are derived from the nearby Eel River Remote Automated Weather Station (RAWS) and a portable RAWS.

The Eel River RAWS was used hourly for all wind data. The portable RAWS was deployed in the fire area from September 23 to October 1. These stations were used to develop the diurnal cycle of maximum temperature—minimum relative humidity, minimum temperature—maximum relative humidity for the fire.

In the early stages of the fire, perimeter data was estimated visually by aircraft and are therefore sparse and imprecise,. No CBI data sample plots were within these initial fire areas. As the fire increased in size, fire perimeters were determined primarily using helicopter mounted thermal imaging technology. Usually only one perimeter was generated at the end of each flight day. The daily fire perimeter was used as an ignition starting point for FARSITE, and the burn was projected for at least 48 hours. Initially, a 6 hour daily burning period was used since the fire advance was initially slow. This was extended to a 10 hour active burning period by the second week of the fire. Additional ignition was added where perimeter conrol firing operations are known to have been used and actual fire advance was not reasonably predicted by model.

Where the fire was projected to advance, FARSITE predicted the following values for each 30 m x 30 m raster cell: time of fire arrival from run initiation; rate of spread; flame length; fireline intensity: heat per unit area. Raster output from FARSITE was imported to ESRI ArcMap for compilation and analysis. For each overlapping CBI sample site and FARSITE raster cell, the resulting fire behavior values were evaluated against the corresponding fire intensity for the understory, overstory, and combined CBI values using linear regression (Microsoft Excel 2003).

Results and Discussion

The Canoe fire produced a complex mosaic of fire effects, with the majority of the burned area classified as low or low-moderate severity, based on remote and field calibrated data. Results of the remote evaluation (Ikonos imagery and aerial photos) were well correlated with the field established CBI ratings for the overstory, but significantly under-estimated the fire severity observed in the understory. The Canoe Fire had a slow rate of spread, and with the deep fuel beds present, long duration smoldering burning was observed. Some patches of high severity effects were observed along the ridges where fire intensity was the greatest.

Since fire severity was under-estimated in all but the high severity areas using remote sensing, modeling the fire using FARSITE had some potential to provide better prediction for these sites. As applied, FARSITE only modeled the advancing fire front, not the long duration burning following the front's passage.

The Composite Burn Index (CBI) results were found to be unrelated to FARSITE produced fire behavior variables using linear regression analysis. The FARSITE outputs of fireline intensity, flame length, heat per unit area, rate of spread, and reaction intensity were poor predictors ($r^2 < 0.10$) and not significant for field derived understory, overstory, and combined CBI values.

Knowing that the longer these models project into the future, the more inaccurate they become, we reassessed our data to use only those CBI plots where the fire arrived within 48 hours, then 24 hours of the run initiation. The linear regression fit did not improve substantially. Review of scatter plot diagrams did not suggest improvement by using transformation functions (Figure 1).

Finding the fire behavior outputs as unrelated to the CBI results is understandable because the basic FARSITE landscape data lacked fuel load information for post-combustion analysis. The fire burned for a long time after the passing of the fire front, which we were unable to model. FARSITE was a useful projection tool for perimeter advance and flame lengths associated with the fire front.

Several additional factors contributed to the poor correlation of fire intensity predictions to field observations. Fire perimeters were usually determined between 1900 and 2100 hours for any given day, generally near the end of the active burning period. The next day's projected progression did not begin until 1100, about 14 hours after the last known fire location. In this area, two

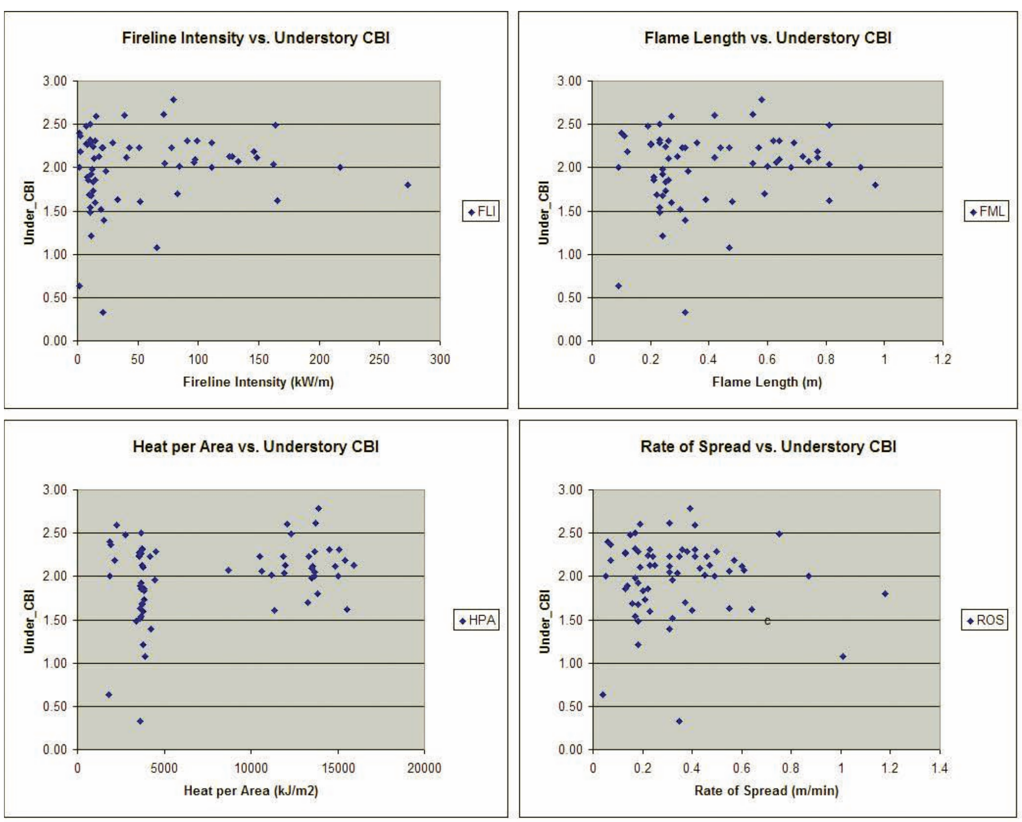

Figure 1—Scatter plot diagrams of fire behavior output versus understory CBI values.

separate burning periods were observed – one during peak fire conditions, and a second beginning at 0100 hours for the upper slopes. The FARSITE simulator is not designed to handle a two-burning period situation, since it relatively uncommon.

Fire control actions also influenced the burn response. In most areas, control lines were established, followed by a firing operation to blacken in the perimeter prior to the arrival of the main fire. We attempted to include these operations in the modeling. However, the records were sparse for when and where these actions were taken and may not have been applied at the correct time or date. Aerial ignition spheres were also used in the fire control operation to accelerate interior burn out in some areas. Higher severity was observed in some of these areas (southeastern portion of the fire) than were predicted by the model.

Differences in winds were not likely a major factor. The dense canopy cover tends to reduce the wind effect in most burn areas. Winds only had substantial effects on exposed ridgelines. Those areas were not used in the CBI assessment. Other error may have been introduced in determining and mapping CBI plot locations (plots landing in the wrong raster cell), and inaccurate assessment of fuel models. However, fuels, topography, and weather did not vary substantially within the immediate area of a plot in either the field, or as modeled. Post-fire vegetation was assessed in the FIREMON process, challenging the accuracy of the remotely sensed fuel model data.

Conclusions

With improved pre-fire data we believe that FARSITE could assist in predicting the landscape effects of fire. Additional research and fuel load data is needed to produce better modeling. Users are cautioned to have a good understanding of model limitations before applying the results. Predicting understory impacts of fire across large areas will remain a challenge without improved remote sensing techniques.

Acknowledgments

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Microsoft Excel 2003 Statistical functions.

Measuring Ecological Effects of Prescribed Fire Using Birds as Indicators of Forest Conditions

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Abstract—*To evaluate the ecological effects of prescribed fire, bird and vegetation surveys were conducted in four study areas of the Klamath National Forest where prescribed fires are being used for management. Bird and vegetation data were collected at sites treated with prescribed fire and nearby untreated control sites. Data were collected at stations from 2000 (pre-treatment) to 2004 (1-4 years post treatment). The treated sites ranged from 9 to 30 ha, and during the course of the study 25-73% of each area was treated with prescribed fire. Over this time period, there was no consistent change in the volume of vegetation in either the tree or shrub strata. Similarly, there was no measurable effect of prescribed burning on the composition of the overall bird community. Spatial variation and annual variation in abundance appear to be more important than the change induced by prescribed burning at this scale and intensity. The abundance of eight individual species that have been identified as conservation focal species for coniferous forests was also investigated. There were no consistent changes in the abundance of these species that we could attribute to the application of prescribed fire. These results suggest that the prescribed fire applied in these treatment units had negligible effects on landbird community composition.*

Introduction

Biodiversity and ecosystem function may be closely linked to historical fire regimes. These regimes have been altered by fire suppression policies implemented in the 20th century (Agee 1993). In an attempt to restore fuel conditions created by historical fire regimes (i.e. mixed-severity; Huff and others 2005), management agencies are using prescribed burns and mechanical fuels treatments that mimic the effects of natural fire. However, the ability of these management activities to mimic the effects of natural fire on habitat structure and animal populations is not well understood (Tiedemann and others 2000). For example, prescribed fire treatments may fail to create the range of habitat conditions used by birds after naturally occurring wildfires (Smucker and others 2005).

Like many national forests across the west, the Klamath National Forest in northern California is currently using prescribed fire as a tool to reduce fuels and improve forest health (S. Cuenca, personal communication). However, the ability of prescribed fire to achieve the desired ecological effects is largely uninvestigated (Tiedeman and others 2000; Huff and others 2005). Monitoring is essential to evaluate the ability of fire-related management activities to achieve desired ecological conditions (Huff and others 2005). One approach to designing monitoring projects is to focus on groups of organisms that can provide cost-effective information about ecological conditions of interest

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(Vos and others 2000; Gram and others 2001). Birds are an effective tool for monitoring because: (1) many species are easily and inexpensively detected using standardized sampling protocols; (2) species respond to a wide variety of habitat conditions; and (3) accounting for and maintaining many species with different ecological requirements can be used to implement landscape scale conservation strategies (Hutto 1998). Changes in the abundance of bird species associated with desired habitat conditions can thus be used to gauge the ability of management actions to maintain or improve that habitat condition and provide inferences about which habitat conditions are contributing to these changes.

To evaluate the impacts of prescribed burning in the Klamath National Forest, we compared vegetation structure and bird abundance over a five-year period. The objectives of this project were to (1) describe the effects of prescribed burning on vegetation structure and bird community composition and (2) evaluate if these effects are consistent with the ecological goals of coniferous forest management.

Methods

Study Sites and Sampling Design

Our study site was on the Klamath National Forest in northern California (fig. 1). The forest vegetation in the area of these prescribed fires is diverse (Whittaker 1960) and includes both conifer and hardwood species. Dominant conifers include Douglas-fir (*Pseudotsuga menziesii*), ponderosa pine (*Pinus ponderosa*), incense-cedar (*Calocedrus decurrens*), and white fir (*Abies concolor*). Dominant hardwoods include tanoak (*Lithocarpus densiflorus*), Pacific madrone (*Arbutus menziesii*), canyon live oak (*Quercus chrysolepis*), California black oak (*Q. kellogii*), Oregon white oak (*Q. garryana*), and big-leaf maple (*Acer macrophyllum*). The relative composition of these species varies with elevation, aspect, and soils. Generally, these forests correspond to the Douglas-fir, Mixed Evergreen Hardwood, or White Fir Types described by Huff and others (2005). Fire-related studies in these vegetation types show a mix of fire severities, frequencies, and sizes typically characteristic of low and moderate-severity fire regimes (Agee 1991; Wills and Stuart 1994; Taylor and Skinner 1998, 2003). Over time, such mixed-severity fires create forests with multiple age classes, often with Douglas-fir or ponderosa pine as an emergent canopy above various hardwoods.

Working with a fire planner and district biologist form the Klamath National Forest, we identified four study areas where a series of control burns were to be implemented (fig. 1). Using maps of planned prescribed fire treatments, we established groups of stations (sites) where fire treatments were planned (treated sites), and where they were at least 1000 m from where fires were planned (control sites). Stations were established at least 250 m apart. For all analyses we consider sites as independent replicates and generated a single measurement for each site by averaging across stations.

The application of prescribed burns within the study areas was patchy. Sometimes, burns were applied such that stations were located along their edges or just outside the boundaries of burns. As a result, it is difficult to use a simple dichotomous classification of treated vs. untreated stations. Furthermore, stations were surveyed each year, but between surveys new treatments were applied. As a result the proportion of treated area around the points increased throughout the course of the study. To quantify the proportion of

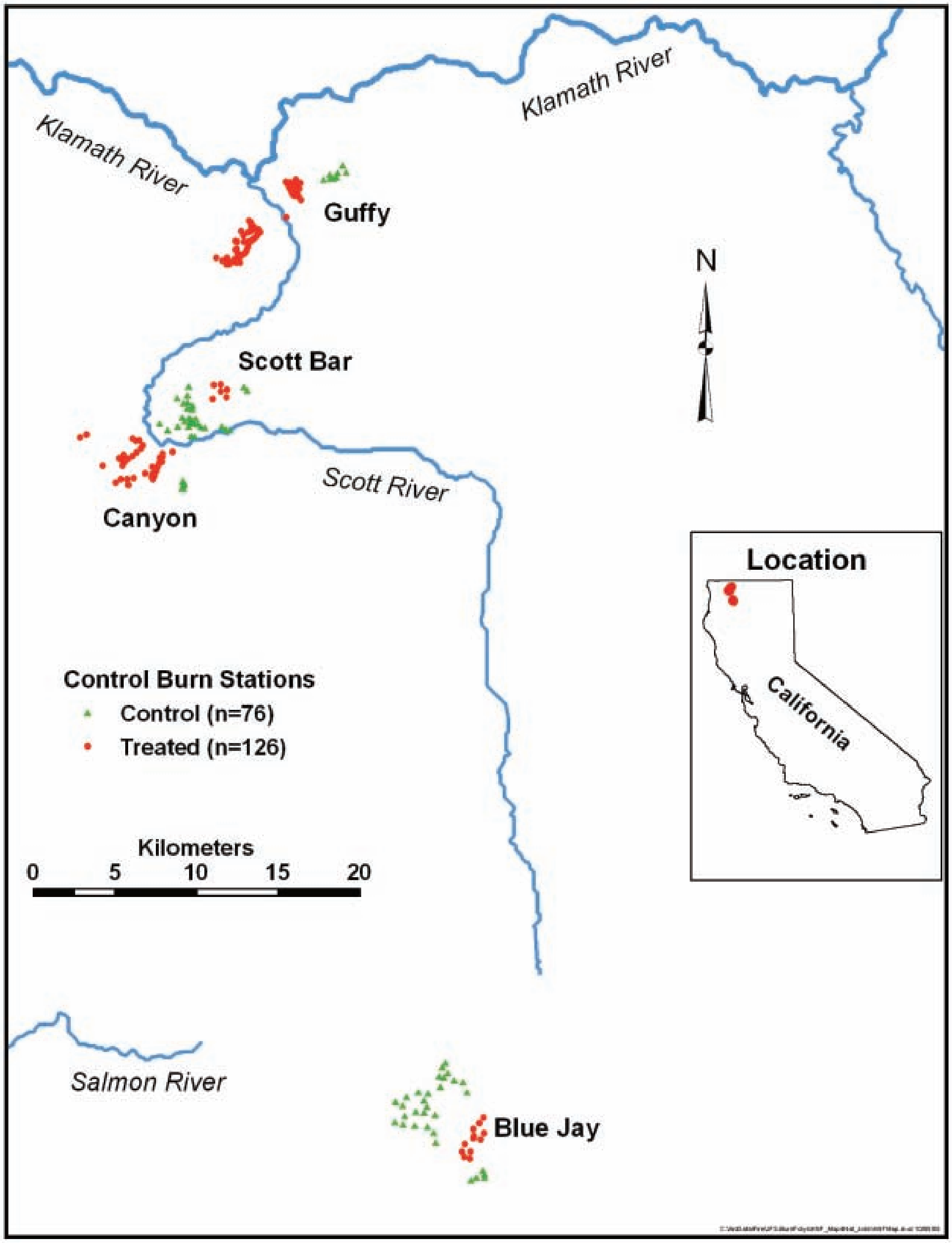

Figure 1—Map showing the location of four study areas where we studied the effects of prescribed fire on bird communities in the Klamath National Forest in northern California. Triangles represent stations at treated sites, and circles represent stations at control sites.

each treated site that was burned, we used a geographical information system to create a 50 m buffer around all points that fell within 50 m of a polygons that had been treated between 1999 and 2004 and then calculated the percent of this area that was treated in each year of the study (table 1).

Data Collection

Vegetation sampling—Vegetation structure was measured at all stations in all years of the study. We used a relevé method (Ralph and others 1993) to collect vegetation data at each station on variable radius plots. Within these

Table 1—Four study units in then Klamath National Forest, California, where prescribed burning was applied between 2000 and 2004. Location of sites are identified in Figure 1.

Area	Site	Number of stations	Total area (ha)[1]	Percent treated[2]				
				2000	2001	2002	2003	2004
Blue Jay	treated	12	9	0	0	18	25	25
	control	31						
Scott Bar	treated	6	5	0	0	0	0	71
	control	8						
McGuffy	treated	69	53	33	33	33	33	59
	control	8						
Canyon	treated	39	30	53	62	64	66	73
	control	29						

[1]Number of ha encompassed by a 50 m buffer around the points in each unit.
[2]Cumulative percent of the buffer-defined area that was treated for each year.

plots, we recognized two vegetation layers: a tree layer (generally >5 m), shrub layer (generally >0.5 m and <5 m). For each layer, we visually estimated height of the top of the tree layer (canopy height) and the bottom of the tree layer (canopy base height). We also estimated shrub height and shrub base height. For each layer, we recorded total cover of all vegetation in each layer as one of six cover classes (0, 0 to 5, 5 to 25, 25 to 50, 50 to 75, and 75 to 100 percent) and used the center point of each cover class as the measurements.

Breeding season point counts—Bird abundance was evaluated using standardized point count methodologies (Ralph and others 1993). Five-minute bird counts were conducted between sunrise and 1000 PDT on each station, and all landbird species seen and heard were recorded. The distance to each individual was estimated to the nearest meter. Counts were conducted only on days when the wind was <20 kph and it was not raining. All observers were experienced and had been trained for distance estimation and species identification. Only birds detected ≤50 m of each point were used in the analysis. This criterion was chosen to reduce the possibility of double counting individuals, including detections that were outside of treated or control areas, and alleviate biases introduced if detection rates differed between treated and control areas (Schieck 1997; Siegel and DeSante, 2003). Flyover detections were excluded from the analysis. We restricted our analysis to passerines and woodpeckers, and excluded four species (Common Raven, American Dipper, Violet-green Swallow, and American Crow) that we expected would be highly influenced by habitat characteristics unaffected by prescribed fire.

Data Analyses

Vegetation structure—We used the relevé data to generate indices that represented the volume of vegetation of the tree layer and shrub layer. The volume of the tree layer was calculated by subtracting the canopy base height from the canopy height, and then multiplying this distance by the total cover value for the tree layer. The same method was used to calculate an index for the volume of the shrub layer. Within each year, we averaged all measurements within each site, and used this single tree and shrub layer value in all subsequent analyses.

To describe the difference between vegetation volume of treatment and control sites, we used:

$$d = \log(V_{treatment}/V_{control}),$$

where d describes the difference between the vegetation volume (V) in the control sites and treatment site. When there is no difference between control and treatment sites $d = 0$, when treatment sites have greater vegetation volume than controls, d is positive, when treatment sites have less vegetation volume d is negative. Because prescribed fire was expected to raise the canopy base height and reduce shrub cover, we predicted that d would become increasingly negative over the course of the study.

Bird community composition— For each site and year we calculated average abundance (individuals/station) of all bird species and used these values in a species x site matrix. We then tracked the movement of each site in ordination space to evaluate the degree to which the bird community composition changed over the course of the study. Because our four areas covered a wide range of elevations and habitats, we expected substantial spatial differences in bird community composition. Therefore, we analyzed two sets of birds; 'all birds' included all the passerines and woodpeckers that were detected during the study and 'core birds,' which was a subset that was restricted to species that were detected at all sites in at least one year of the study. We evaluated changes in bird community composition through time using detrended correspondence analysis (DCA) conducted in PC-ORD (McCune and Mefford 1999).

Abundance of coniferous forest focal species—To investigate species-specific responses to fuels treatments we selected 'core' birds that were identified by either the California or Oregon/Washington Partners in Flight coniferous forest conservation plans (Altman 2000; CalPIF 2002). Within each year, we averaged the number of individuals detected per station, and used this single value for each site in all subsequent analyses. Similar to the analysis of vegetation volume, we described the difference in bird abundance between treated and untreated sites as:

$$d = \log(A_{treatment} + 1/A_{control} + 1),$$

where d describes the difference between bird abundance (A) at control sites and treated sites. Because some species were not detected at some sites in some years, we used Naperian (N + 1) logarithms.

Results

Application of Prescribed Fire

Prescribed fires were applied at all four sites over the five years of the study (table 1). At two sites (Guffy and Canyon) a third to half of the area had already been treated before the study began, however, in both these areas treatments continued throughout the course of the study (table 1), thus we would expect the trajectory of changes at these areas to be similar to the other areas. In most of the sites we monitored for several years after the first treatments were applied, with the exception of the Scott Bar site, where we collected a single year of post-fire data.

Vegetation Structure

We found no evidence that the volume of live vegetation in the tree layer was consistently reduced at treated sites; in each year the difference between the treated and control areas was roughly symmetrical around 0, and there was no suggestion that this measurement had consistently decreased at any of the four areas (fig. 2). Our results for the volume of the shrub layer were similar (fig. 2), in that there were no sites that showed a consistent pattern of change between treated and control sites through the course of the study. In both the first and last year of the study, the measurements of the difference in total shrub cover of treated and untreated sites was symmetrically distributed around 0 (fig. 2).

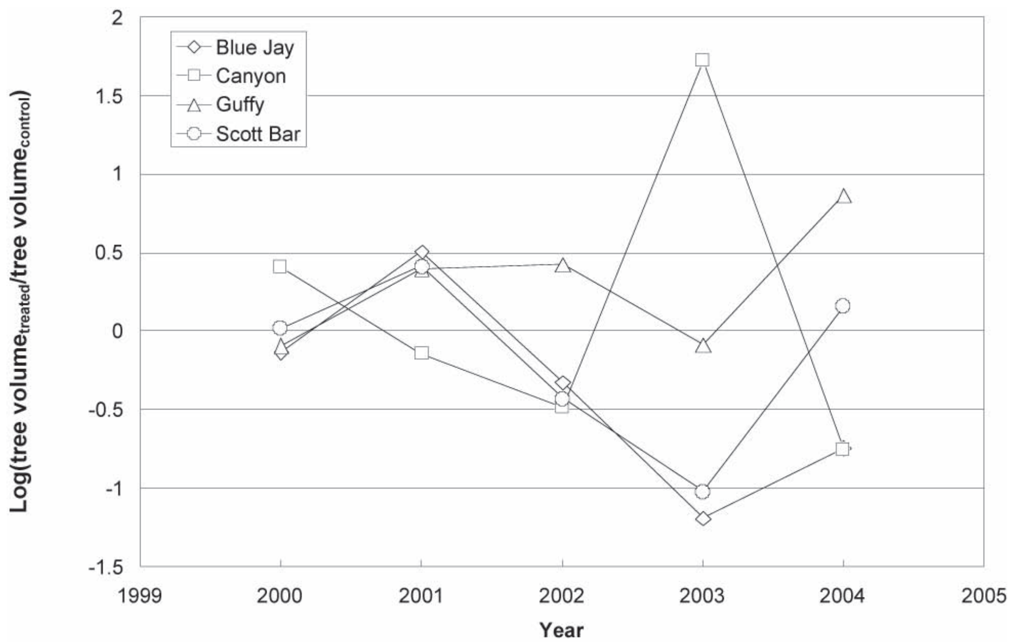

Figure 2—Log response ratios comparing vegetation characteristics of treated and control sites from the four study areas over the five-year study period.

Bird Community Composition

For ordinations of both 'all birds' and 'core birds' most of the variation in the original multidimensional space was captured in the first two axes (table 2), therefore, we limited out our interpretation to these axes. Ordination of bird communities for the treated and untreated units demonstrated substantial variation in bird communities among sites (fig. 3). In particular, the Canyon control site and Blue Jay treated site were substantially different from all the other study sites. Furthermore, it was not uncommon for sites from the same area (e.g., compare Guffy treatment to Guffy control) to be more different than sites from different areas (e.g., Guffy treatment versus Scott Bar control). These spatial patterns remained roughly the same for ordinations of all birds and core birds (fig. 3). Although there was substantial year to year variation in bird communities, both in treated and control units, there was no apparent directional movement in ordination space associated with treatments. For instance, although treated units Canyon and Blue Jay both moved during the study period, they moved toward each other, suggesting that if there was an effect of prescribed fire, it had the opposite effect in these two units.

Abundance of Focal Species

For the eight Partners in Flight coniferous forest focal species that we investigated, we could discern no obvious changes in abundance that occurred as a result of treatment (fig. 4).

Discussion

Our results suggest that the effects of prescribed fire on vegetation structure and bird community composition have been minimal in these areas of the Klamath National Forest. We found no evidence that prescribed fire treatments were associated with a persistent decrease in the volume of vegetation in the tree or shrub layer. There was substantial year to year variation, and some of these changes may represent short term changes from recent treatments, but these effects did not appear to persist, or accumulate, over the course of the study.

Similarly, our ordination results for the bird community show no evidence of a directional change in bird community composition that is unique to the treated areas (fig. 3). Even in the absence of overall community effects, we

Table 2—Coefficient of determination for the correlation between bird community detrended correspondence analysis (DCA) ordination distances and relative Euclidean distances in the original multidimensional space.

	All birds		Core birds	
DCA Axis	**Incremental R^2**	**Cumulative R^2**	**Incremental R^2**	**Cumulative R^2**
Axis 1	0.39	0.39	0.39	0.39
Axis 2	0.35	0.74	0.40	0.79
Axis 3	0.04	0.79	0.04	0.83

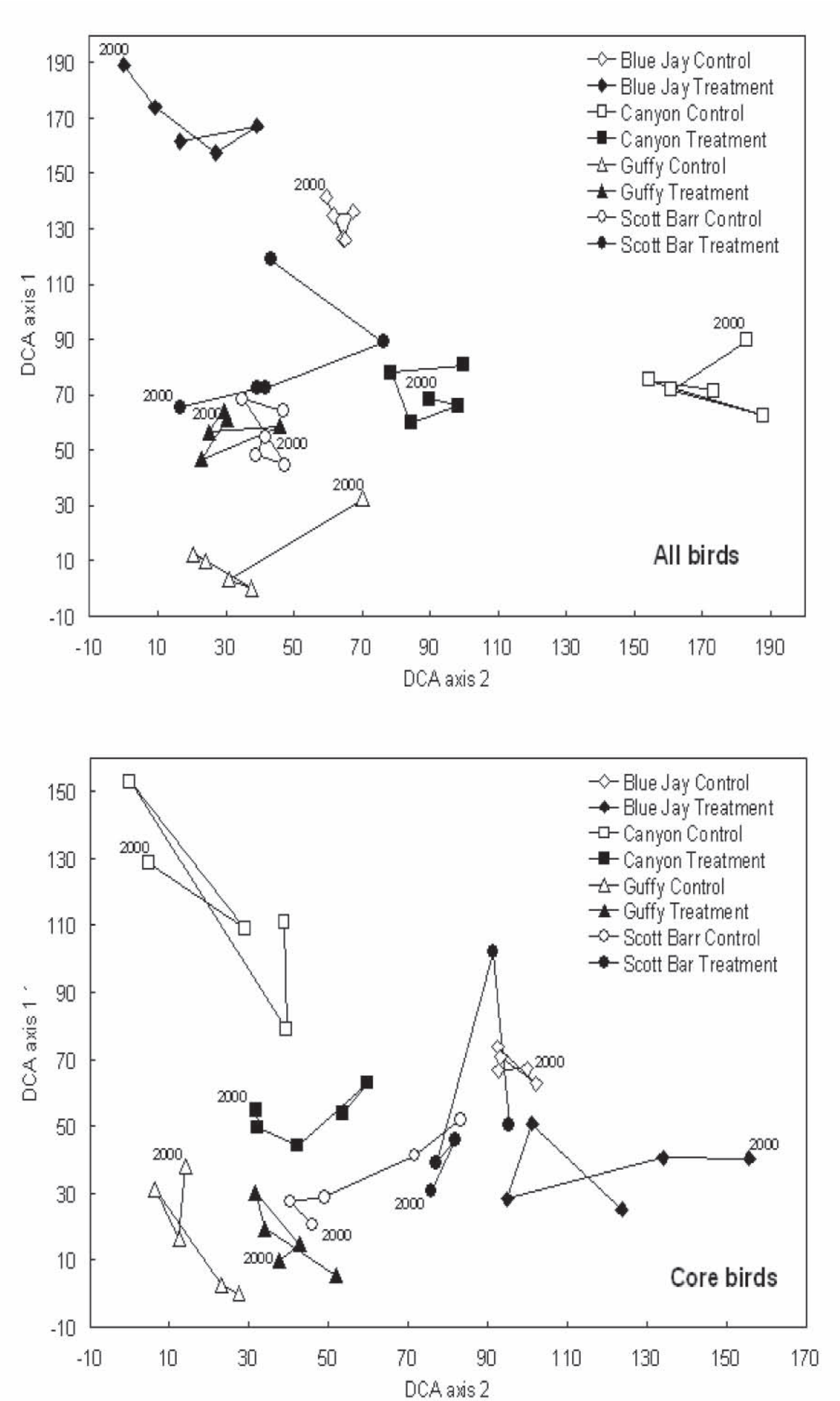

Figure 3—Ordination plots of DCA scores for bird communities at treated and untreated sites in the Klamath National Forest in northern California.

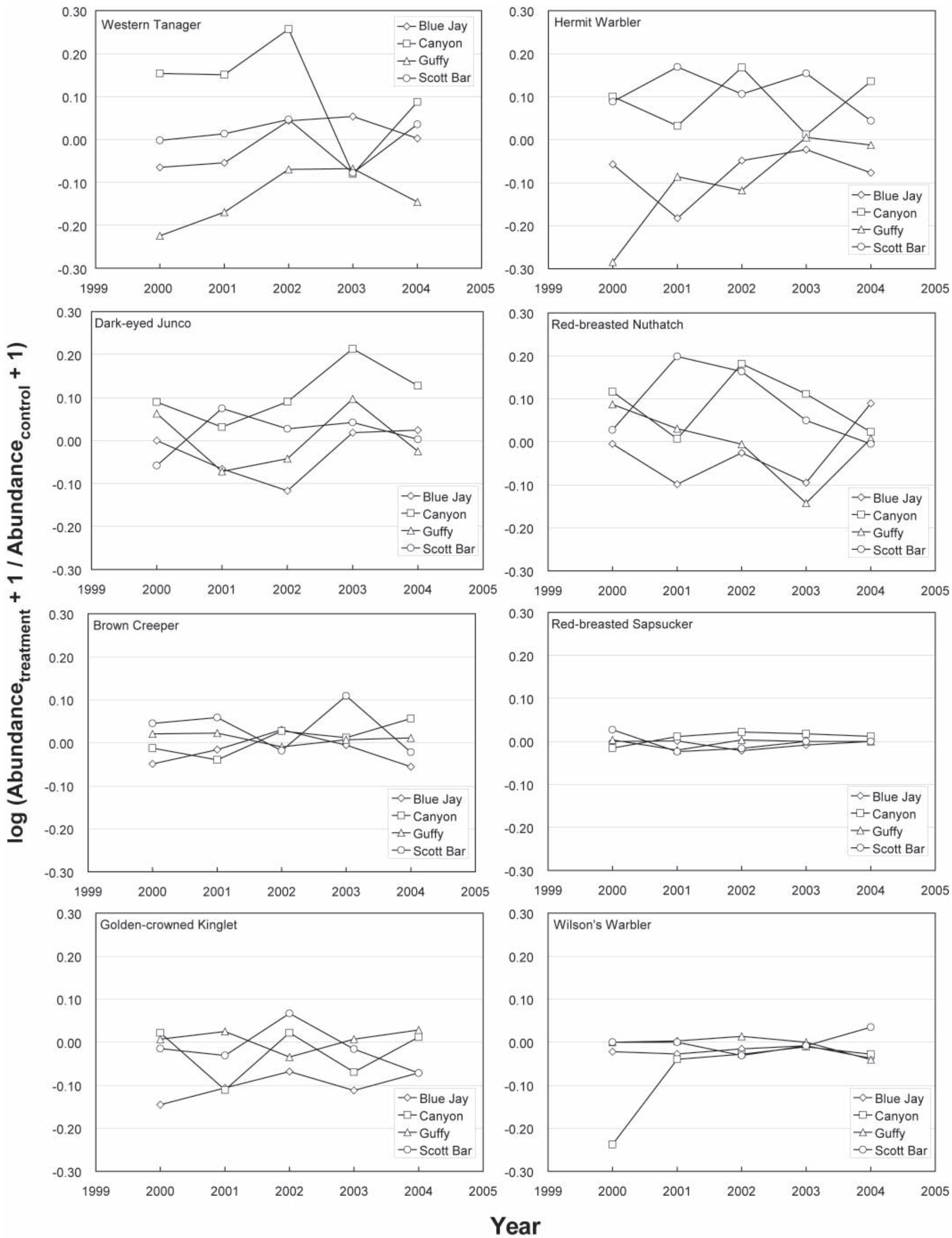

Figure 4—Log response ratios comparing bird abundance of treated and untreated sites from the four study areas over the fire-year study period.

may still be concerned about the effects of prescribed fire if they change the abundance of individual species that are of particular conservation concern. However, our analyses of the Partners in Flight focal species for coniferous forests showed no consistent trends for these species to become either more or less abundant after treatment.

There is limited evidence that fuels reduction projects in the western United States can be implemented in such a way that they are consistent with the goals of wildlife conservation and ecosystem health (Tiedemann and others 2000; Huff and others 2005). However, this study, and a similar study comparing thinned and unthinned mixed-conifer forests in the Sierra Nevada (Siegel and DeSante 2003), suggest that in conditions were prescribed fire has little effect on the volume of live vegetation, such treatments may have relatively minor consequences for bird communities. However, if the goal of these treatments includes restoring conditions in such a way that it changes the quality of wildlife habitat, our results suggest that prescribed fire in the Klamath National Forest would need to be modified to achieve the desired conditions.

Acknowledgments

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Monitoring Changes in Soil Quality from Post-fire Logging in the Inland Northwest

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Abstract*—The wildland fires of 2000, 2002, and 2003 created many opportunities to conduct post-fire logging operations in the Inland Northwest. Relatively little information is available on the impact of post-fire logging on long-term soil productivity or on the best method for monitoring these changes. We present a USDA Forest Service Northern Region study of post-fire logged sites using a variety of methods to assess changes in soil productivity and site sustainability after timber harvesting activities. The disparate soil and climatic conditions throughout the Northern Region made it an ideal area to study post-fire logging operations. Our results indicate that post-fire logging during the summer creates more detrimental disturbance (50% of the stands) than winter harvesting (0% of the stands). In addition, on the sites we sampled, equipment type (tractor > forwarder > rubber-tired skidder) also influenced the amount of detrimental disturbance. Number of sample points is a critical factor when determining the extent of detrimental disturbance across a burned and harvested unit. We recommend between 80 and 200 visual classification sample points, depending on confidence level. We also provide a summary of methods that will lead to a consistent approach to provide reliable measures of detrimental soil disturbance.*

Introduction

During the last century, wildfires in the western USA have been viewed by many land managers and the public as catastrophic events (Kuuluvainen 2002). Until recently, fire suppression has been used to control the extent of these fires, but now stand-replacing fires are occurring on many Federal lands in the western USA. Consequently, the standard policy on many National Forests has been to harvest fire-killed trees for economic value before they decay (Lowell and Cahill 1996; McIver and Starr 2001). Proponents and opponents of post-fire logging are abundant (Beschta and others 2004; Sessions and others 2004; Donato and others 2006), but one critical issue of concern to each group is the impact of this practice on the soil resource.

Wildland fires can impact more than 10,000 ha of forest land at one time and, combined with post-fire logging, significant soil impacts can occur. Loss of surface organic matter and nutrients from the fire, increased decomposition from increased insolation, decreased soil porosity, increased erosion, and compaction may all combine to alter site productivity after wildfire and post-fire logging activities (Poff 1996). There are no specific methods that directly assess the impact of post-fire logging on soil productivity, but many methods for measuring proxies exist (see Burger and Kelting 1999; Schoenholtz and other 2000). Measures of wood production, net primary productivity, or changes in some specific soil properties (e.g. bulk density, forest floor depth,

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cover type, etc.) can all be readily determined, but the link between forest management, soil properties, and site sustainability is not easily obtained.

Historically, maintenance of soil productivity on public lands in the USA has been governed by the Multiple Use Sustained Yield Act of 1960, the Forest and Rangeland Renewable Resources Planning Act of 1974, and the National Forest Management Act of 1976. As an outgrowth of these policies, each USDA Forest Service Region developed soil quality standards and guidelines, which were designed to act as a first warning of reduced site productivity after harvest and site preparation operations. The general concepts and the basis for the various guidelines are described in Griffith and others (1992). Lacking better methods, these standards and guidelines have also been used to evaluate soil productivity changes after wildfire and post-fire logging.

Concern about an accurate assessment of soil properties has expanded because of the growing public interest in the consequences of forest management practices on soil quality and its productive capacity (Burger and Kelting 1999; Schoenholtz and others 2000). Worldwide initiatives including the Helsinki Process (1994) and the Montreal Process (1995) have resulted in the development of criteria and indicators for monitoring sustainable forestry practices at broad levels (Burger and Kelting 1999). Recently, progress has been made on developing a common approach to soil monitoring in northwestern North America (Curran and others 2005). The key questions are: What do we measure and what does it mean? The literature is rife with examples of how a soil chemical, physical or biological property may contribute to changes in biomass production, hydrologic function, or ecosystem sustainability (see Schoenholtz and others 2000 for a summary). However, as budgets and personnel dwindle, land managers need a visual assessment of disturbance that can be completed quickly, efficiently, and easily by either field soil scientists or others trained in the assessment process (Curran and others 2005).

Wildfires and post-fire logging generate unique soil surface conditions. Visual disturbance criteria estimate the amount of detrimental disturbance and may need to be specifically designed to encompass the impacts of both fire and logging. Therefore, the objectives of our study were to: (1) determine the magnitude and areal extent (as defined by current soil quality standards) of detrimental disturbance from wildfire and post-fire logging across the Northern Region of the USDA Forest Service, (2) determine the most appropriate spatial sampling design methods for assessing the magnitude of soil impacts, and (3) develop visual criteria that can be used following post-fire salvage harvests to assess disturbance across disparate soil and climatic regimes.

Methods and Materials

Site Descriptions

In the summer of 2004 and 2005, post-fire logging sites were located on the Custer, Helena, Bitterroot, Kootenai, Lewis and Clark, Flathead, and Lolo National Forests (Table 1). Thirty-six stands were sampled over 2 field seasons; 20 had been post-fire winter logged and 16 were post-fire summer logged. Sites were selected by local soil scientists in areas that had recently burned in a wildfire (2000, 2002, or 2003) and had subsequently been logged. If available, we selected three replicate units on each forest, which had similar slope, aspect, soil type, and logging practices.

Table 1—Post-fire logging study site characteristics.

Season of harvest	Logging method	National Forest	Year burned	Year of harvest	Elevation (m)	Parent material	Surface soil texture
Summer	Tractor	Custer	2002	2003	1200	Sandstone	Loamy sand
	Tractor	Helena	2000	2003	1700	Metasediments	Sandy loam
	Tractor	Helena	2000	2002	1700	Metasediments	Loamy sand
	Forwarder	Lolo	2000	2005	1400	Metasediments	Loamy sand
	Forwarder	Flathead	2000	2002	1900	Quartzite	Sandy loam
Winter	Tractor	Bitterroot	2000	2002	1750	Granitic	Loamy sand
	Tractor/RTS[1]	Flathead	2000	2002	1150	Limestone	Silt loam
	Tractor	Helena	2000	2002/03	2500	Metasediments	Sandy loam
	Tractor	Helena	2000	2002	1700	Metasediments	Loamy sand
	Tractor	Lewis & Clark	2001	2003	2200	Limestone	Silt loam
	Forwarder	Kootenai	2000	2003	1600	Glacial till	Silt loam
	Forwarder	Lolo	2000	2005	1500	Metasediments	Loamy sand

[1] RTS= Rubber tired skidder.

Soil Indicator Assessment

In each post-fire logging unit, a 100 point systematic grid and a 100 point random transect were established from a fixed corner point. At each grid and transect point, we described the soil surface cover (e.g. rill erosion, forest floor, bare mineral soil, rocks, etc.) and the presence or absence of platy structure in the underlying mineral soil in 1 m^2 plots. Once the soil surface had been described, we assigned a soil disturbance category to each plot (Table 2), based on the classification systems of Howes (2001) and Heninger and others (2002). In addition to a visual classification, soil strength was determined at each sampling point using a RIMIK CP40 recording penetrometer (Agridry, Toowoomba, Australia).

Statistical Analysis

Chi-square tests for homogeneity were used to evaluate the relationships between disturbance class and soil texture, parent material, season of harvest, and harvest method. Chi-square tests for homogeneity were also used to evaluate relationships between detrimental soil disturbance, soil texture, parent material, season of harvest, and harvest method. Analysis of variance was used to examine relationships between soil strength and soil texture, parent material, season of harvest, and harvest method. All analyses were performed using SAS 9.1.

Results

In this study, there were no significant differences between the grid and random transect methods when visually assessing soil disturbance after fire and post-fire logging ($p < 0.001$). Therefore, data from both the grid and random transect were pooled for subsequent analyses.

Table 2—Description of soil condition classes used.

Condition class	Identifying features
0	Undisturbed forest floor
1	No evidence of past equipment operation, but records of harvesting No wheel ruts Forest floor intact No mineral soil displacement
2	Trail used by harvester (ghost trails) Faint wheel tracks and ruts Forest floor intact No mineral soil displacement and minimal mixing with forest floor
3	Trail used by harvester and forwarder Two track trails created by one or more passes Wheel tracks are >10 cm deep Forest floor is missing/partially intact
4	Skid trails existed prior to reentry and reused Old skid trails from 20th century selective harvest Recent operation had little impact on old skid trail Trails have a high level of soil compaction Evidence of mineral soil displacement from trails
5	Old and new skid trails present Mineral soil displacement from area between skid trails Forest floor is missing

In the USDA Forest Service Northern Region, a stand is considered detrimentally disturbed if greater than 15% of the area is in disturbance class 3, 4, or 5 (Table 2). Of the stands we sampled, 50% of the summer-logged sites and no winter-logged sites had more than 15% of the sampling points in the detrimental disturbance categories (Table 3). The relationship of logging season and detrimental disturbance is significant ($p < 0.0001$) and is primarily characterized by platy structure on skid trails or cow trails.

Table 3—Average soil disturbance after summer and winter post-fire logging.

Season of harvest	National Forest	Number of stands	Amount of Disturbance	
			Not detrimental	Detrimental
			percent	
Summer	Custer	4	72	28
	Flathead	3	77	23
	Helena	3	96	4
	Lolo	4	91	9
Average			**84**	**16**
Winter	Bitterroot	3	97	3
	Flathead	3	90	10
	Kootenai	3	97	3
	Lewis & Clark	3	92	8
	Lolo	2	99	1
	Helena 1	3	92	8
	Helena 2	3	87	13
Average			**93**	**7**

There is a significant relationship ($p<0.0001$) between site parent material and the areal extent of detrimental disturbance. Metasediments, limestone, and granitic parent materials were the least detrimentally disturbed with 75% of the visual classification points being in class 0 or 1.

Surface soil strength was generally not related to disturbance class; however, some exceptions occurred at the 2.5 cm depth. The exceptions were two stands on the Helena National Forest ($p = 0.0312$; $p = 0.0236$) and two stands on the Flathead National Forest ($p = 0.0235$; $p = 0.0033$). These four stands are unique as there was no relationship between surface soil strength, harvest season, type of equipment, or total areal extent of disturbance. However, all four of these sites were burned in 2000 and post-fire logged in 2002. The time between post-fire logging and sampling could have been enough for some soil recovery before soil monitoring occurred.

For all sites, there is a significant relationship ($p < 0.0001$) between visual disturbance class, areal extent of detrimental disturbance, and harvest method. In 66% of the forwarder harvested units, 85% of the rubber-tired skidder units, and 45% of the tractor units, we detected less than 15% areal extent of detrimental disturbance. Many of the sampling sites classified as not detrimentally disturbed had less exposed bare mineral soil than detrimentally disturbed units ($p < 0.0001$). On sites with a significant portion of soil cover, many had live plants, forest floor, moss and lichens present, which may likely indicate soil surface recovery after post-fire harvesting.

Discussion

Severe wildfires greatly impact below-ground ecosystems, including development of water-repellent soils (DeBano 2000) and decreased evapotranspiration (Walsh and others 1992), which can lead to overland flow of water and significant soil erosion. Additionally, the loss of forest floor material reduces water storage in the surface mineral soil (McIver and Starr 2001). The subsequent cumulative effects of fire followed by logging in such a landscape have been difficult to measure (McIver and Starr 2001). Soil surface conditions after post-fire logging is highly influenced by management decisions, which determine equipment type and harvest season. Regardless of disturbance origin (fire or logging), soil productivity in a given area may be influenced by site characteristics (topography, parent material, revegetation, and climate), logging method, and construction of additional roads or skid trails. Our visual disturbance classes (0-5) along with a quick presence or absence survey of key factors (platy or massive structure, forest floor displacement, rut, sheet, rill, or gully erosion, mass movement, live plant, forest floor, wood debris <3″ or >3″, or bare soil) can determine if a harvest unit will meet soil quality guidelines. However, our disturbance classes need to be modified to include soil burn impacts associated with severe wildfires. Removal of surface organic matter may not be detrimental to site productivity unless it is coupled with a change in color in the mineral soil (Neary and others 1999).

Detrimental disturbance was least with rubber-tired skidders, greater when using forwarders, and the most with tractors. In addition, the number of stands with detrimental disturbance was significantly decreased when logging operations occurred during the winter. This is similar to work by Klock (1975) in which he found that tractor skidding over exposed mineral soil caused the greatest amount of detrimental disturbance (36%), followed by cable skidding (32%), and tractor skidding over snow (10%).

Eighty-two percent of our stands were categorized as not having a detrimental soil disturbance after post-fire logging. The remaining stands that approached or exceeded the 15% areal extent of detrimental soil disturbance may require amelioration before other management activities are considered. Detrimental soil disturbance ratings are generally higher after wildfire and post-fire logging when compared to green timber sales, since both wildfire and post-fire logging sites generally lack understory vegetation and forest floor (Klock 1975). Ground-based logging can mitigate some detrimental impacts by leaving logging residue on site or by delaying harvesting until after killed trees drop their needles after a wildfire to establish some forest floor. Both measures provide additional protection from erosion (Megahan and Molitor 1975).

Compaction of the surface soil is also a common concern after ground-based logging operations (Froehlich 1978; Adams and Froehlich 1981; Clayton and others 1987; Page-Dumroese 1993; Miller and others 1996), and surface soil disturbance is more evident immediately post-harvest. Using visual classification categories, we were able to distinguish impacts of summer and winter logging, the influence of parent material, and harvest methods. In some cases, our visual assessments were a direct indication of changes in soil physical properties (e.g. platy or structure) or in surface properties (e.g. displacement of surface organic matter, churned mineral soil, or ruts), and could be used as a surrogate for more intensive sampling. However, the time elapsed between the wildfire and logging activities, and the time between post-fire logging and soil monitoring can be important factors in the degree of detrimental disturbance measured. For instance, on sites with several years between the fire and logging and then another time period between logging and monitoring, some revegetation would likely occur and deposit plant litter on the soil surface. Plant establishment could improve some soil physical properties and influence whether a sample point is categorized as detrimental (class 3) or not detrimental (class 2). The short times between fire, logging and monitoring (1 year between each) may be a reason the Custer National Forest had 28% detrimental disturbance, compared to the Helena National Forest (3 years between fire and logging, and 1 year between logging and monitoring) with only 4% detrimental soil disturbance.

Soil resistance, as measured using a penetrometer, could be easily evaluated on many sites, but the influence of rocks, roots, and low soil moisture, later in the growing season limited its usefulness as tool to make compaction comparisons among sites. However, the use of the penetrometer within one area of similar soil characteristics during a time when soil moisture is fairly high (near field capacity) is feasible for monitoring changes in soil penetration resistance (Utset and Cid 2001).

Management Implications

For our study, we used 6 visual disturbance categories (classes 0-5) to describe areas that had been burned by wildfire and subsequently logged. These visual disturbance classes described combinations of soil disturbance that recur across each harvest unit and can be a relatively quick and easy method for quantifying soil disturbance (Howes et al. 1983). However, season of logging, equipment used, and time between disturbance activities and monitoring were important variables that determine the extent of detrimental disturbance. The visual classification measurements do seem to be an easy, inexpensive method for timely monitoring, and with more data collection, can likely be correlated with long-term vegetation growth. Visual classifications that encompass burn conditions of the soils (charcoal, mineral soil discoloration and ash deposition) are also needed to refine the disturbance assessments, which would make them more useful to forest managers and soil scientists.

Our data indicate that at the 95% confidence level, a sample size of approximately 200 sample points in a 10 ha unit would detect 15% (±5%) detrimental disturbance (Table 4 and unpublished data). A site with 5% detrimental disturbance would only need 75 sample points; whereas a site with a high proportion (>30% of the unit) of detrimental disturbance would need 340 sample points at this confidence level. A confidence level of 80% would significantly lower the number of samples needed. For instance, a site with little disturbance (<5% of the unit) would need only 32 sample points, but a site with a large amount (30% of the unit) of disturbance would need 139 sample points. Using either random transects or grid points are appropriate strategies for laying out monitoring points for similar wildfire burned and post-fire harvested sites when using our visual classification method.

In the USDA Forest Service, soil assessment of management impacts is typically linked to site productivity through soil quality standards (Page-Dumroese and others 2000). However, these standards are not site-specific, do not specify collection of baseline data, are not always linked to changes in biomass production or carbon accumulation, and, in many cases, the monitoring techniques are cumbersome, lengthy, costly and require some laboratory analysis. Reliable assessment of soil disturbance and the link to site productivity is critical. Visual classifications have been used throughout the Pacific Northwest by the B.C. Ministry of Forests (Forest Practices Code Act 1995) and Weyerhaeuser Company (Scott 2000), but have not been linked to tree growth. To date, visual classification systems only describe surface soil conditions, and have not been validated to response variables that are ecologically important (e.g. tree growth, survival). A necessary step in the acceptance of any visual soil disturbance criteria is to develop direct evidence that there is a change in site function, productivity, or sustainability (Curran and others 2005). Our test of visual criteria for assessing soil disturbance after wildfire and logging operations could be used to determine areal extent of detrimental impacts within a harvest unit.

Although visual classifications are not directly linked to ecosystem functions at this time, it is generally recognized in the northwestern USA that surface organic matter can help maintain site productivity (Page-Dumroese and others 2000; Jurgensen and others 1997; Harvey and others 1981).

Table 4—Sample points needed to detect 15% areal extent of detrimental disturbance in a 10 ha unit at different confidence levels (±5%).

Confidence level	Sample points needed
95%	196
90%	139
80%	84

Existing studies such as the North American Long-Term Soil Productivity (LTSP) study, established in the USA and Canada, are investigating the effects of OM removal and compaction on soil productivity (Powers and others 2004), but fire was not included as a disturbance variable. However, the physical removal of surface OM on LTSP study sites generally resulted in lower mineral soil C pools and reduced N availability 10 years after treatment, and tree growth was reduced on low productivity sites (Powers and others 2005). Additionally, tree growth declined on compacted clay soils and increased on sandy soils, but was strongly related to control of the understory vegetation. Recently, the Fire and Fire Surrogate study was started by the USDA/USDI to evaluate the effects of mechanical fuel reduction treatments and prescribed fire-severity on above- and below-ground productivity in a variety of forest ecosystems across the USA (Weatherspoon 2000). Both of these sources of information are needed to complement monitoring data to help develop post-fire harvesting methods that maintain adequate amounts of OM and limit soil compaction to maintain soil productivity.

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The Relation Between Forest Structure and Soil Burn Severity

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Abstract*—A study funded through National Fire Plan evaluates the relation between pre-wildfire forest structure and post-wildfire soil burn severity across three forest types: dry, moist, and cold forests. Over 73 wildfires were sampled in Idaho, Oregon, Montana, Colorado, and Utah, which burned between 2000 and 2003. Because of the study's breadth, the results are applicable for understanding how forest structure relates to post-wildfire soil burn severity within Rocky Mountains forests. This paper discusses a burn severity classification that integrates fire intensity, fire severity, and post wildfire response; and discusses the relations wildfire setting (fire group), tree crown ratio, tree canopy cover, surface fuel condition, and tree size have with different soil burn severity outcomes.*

Introduction

Although canopy bulk density, fuel models, canopy base height, and other forest metrics have been related to fire behavior using physical laws, controlled experiments, and models (Graham and others 2004, Peterson and others 2005), there is limited information to indicate how forest structure influences or is related to burn severity (what is left and its condition) after a wildfire event (Broncano and others 2004, Loehle 2004, Weatherspoon and Skinner 1995). Moreover, the uncertainty of these relations is unknown, preventing forest managers from communicating their confidence in fuel treatments that may reduce the risk of wildfires and their effects. Without these estimates, managers and forest stakeholders could have a false sense of security and a belief that if a wildfire occurs after a fuel treatment the values they cherish (for example, homes, wildlife habitat, community water sources, sense of place) will be protected and maintained both in the short- (months) and long- (10s of years) term.

In 2001, we began to define and quantify the relation between forest structure and soil burn severity and determine the uncertainty of the relations (Jain and Graham 2004). Although other studies have quantified this relationship they often were limited in scope and applicability (Cruz and others 2003, Martinson and Omi 2003). To avoid these shortcomings, we designed our study to sample many different wildfires (73) that burned throughout the inland western United States over multiple years. Because of the study's scope, it incorporated a large amount of variation in forest structure as well as disparity in burn severity after extreme wildfires. The data we collected came from wildfires that burned in the moist, cold, and dry forests between 2000 and 2003. By including wildfires that burned throughout the inland

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western United States occurring over multiple years, we were able to include a variety of weather (that occurred during the fires) and physical settings in our sampling. The relations between forest structure and soil burn severity and the uncertainty of these associations after intense and severe wildfires will provide information that can be used for informing fuel management decisions throughout the moist, cold, and dry forests of the inland western United States.

Methods

We visited 73 areas in Montana, Idaho, Colorado, Oregon, Utah, and Arizona burned by wildfires between 2000 and 2003 (fig. 1). These wildfires occurred in three forest cover types: dry (ponderosa pine, *Pinus ponderosa* and Douglas-fir, *Pseudotsuga menziesii*), moist (western hemlock, *Tsuga heterophylla*, western redcedar, *Thuja plicata*, grand fir, *Abies grandis*, white fir, *Abies concolor*) and cold (lodgepole pine, *Pinus contorta* and subalpine fir, *Abies lasiocarpa*) forests throughout the inland western United States. Since not all forest burned in a single year, we included multiple years and multiple geographic regions in our data collection (fig. 1). All areas were sampled the summer after they burned, except areas in Flathead and Lincoln counties in Montana and the Diamond Peak complex of fires in Idaho, which burned in 2000. These wildfires were sampled the second summer after they burned.

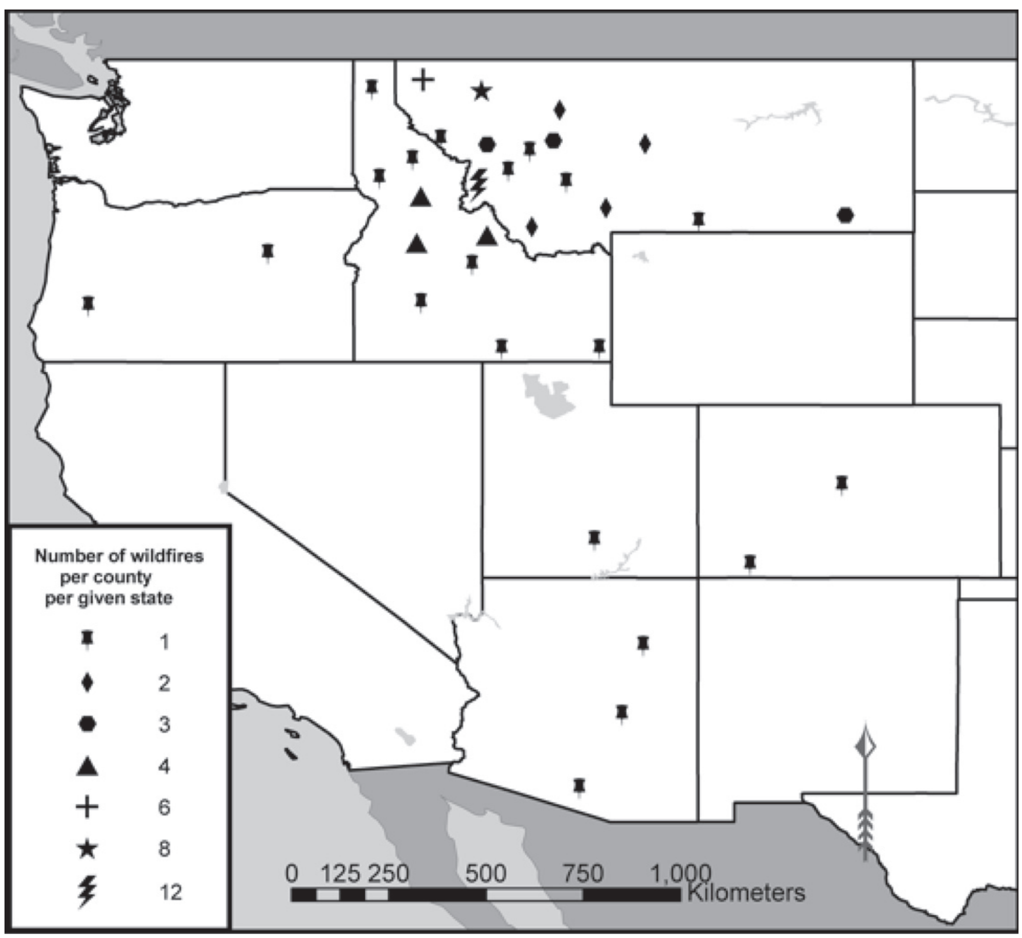

Figure 1—Distribution of the seventy-three wildfires sampled between 2001 and 2004.

Sampling Designs

We used three sampling designs to capture the variation in burn severity occurring at different spatial scales. Intensive sampling occurred in 28 wildfires that burned between 2000 and 2003. Extensive sampling revisited previously established Forest Inventory and Analysis (FIA) plots within 61 wildfires that burned in Montana and Idaho in 2000 and those burned in Montana during 2001 and two wildfires were visited using focused watershed (142 ha to 6,480 ha) sampling.

Intensive Sampling

For each selected wildfire (28 fires), we used stratified random sampling to ensure the variation in forest structure, physical setting, and weather were represented. Our sampling stratification began with forest cover (dry, moist, and cold), followed by burning index (two classes), slope angle (two classes), canopy height (two classes), and stand density (two classes). In establishing the sampling frame, forest cover type described the broad-scale vegetation. We used fire progression maps, local weather data, and the most applicable fuel model for each stand within a fire perimeter to calculate Burning Index (Bradshaw and Britton 2000). We split our sampling at the median burning index for all stands burned by a particular wildfire. The physical settings of the stands were placed into two strata: those with slope angles less than or equal to 35 percent and those with slope angles greater than 35 percent. The Hayman fire in Colorado and Flagtail fire in Oregon had moderately steep topography where we used a 25 percent slope angle to differentiate the two classes. Nested within slope class, stands were divided into sapling to medium sized trees ($\leq$12.5 m) and mature to old trees (>12.5 m). Within height class, two density stratum were identified: those with canopy cover $\leq$35 percent and those with canopy cover >35 percent. All stands within a fire perimeter had an equal probability of being selected. We randomly selected a stand if it 1) met the sampling criteria, 2) had an opportunity to burn, 3) did not have any confounding factors (evidence of suppression activities), and 4) was at least 100 m by 100 m in size.

Extensive Sampling

Interior West Forest Inventory and Analysis staff have randomly located permanent forest sample plots throughout the forests of the western United States. Several of these plots burned in 2000 and 2001 (61 wildfires). Wildfires that burned in Idaho and Montana in 2000, all wildfires that burned in Montana in 2001, and the wildfires that burned in Utah and Arizona in 2003 were revisited. Because FIA plots were distributed across spatially defined grids and the burned areas varied in size and location, the number of plots burned by the fires varied considerably. As a result, some burned areas had multiple FIA plots sampled after a wildfire while other areas only had one plot revisited.

Focused Watershed Sampling

The focused watershed sampling occurred within forests burned by the Quartz and Diamond Peak fire complexes in Idaho and Oregon in 2000 and 2001. Using GIS based maps, we delineated the watersheds burned by these two wildfire events and subsequently defined a 60-m riparian zone along each side of the stream reaches. Areas outside the riparian zone within each watershed were defined as the upland zone. A minimum of twenty-five plots were randomly located within both the upland and riparian zones using a complete spatial randomness (CSR) Poisson process (Diggle 2003). Using this approach, spatial autocorrelation was avoided (Cressie 1991).

Data Collection

Our intention was to develop a continuous variable or post classify the burn severity of the forest floor. To do so, fine resolution descriptors of soil burn severity were synthesized from past burn severity characterizations to develop the burn severity indicators. Our soil burn severity concentrated on what was left after the fire and not what was consumed (DeBano and others 1998, Key and Benson 2001, Ryan and Noste 1985, Wells and others 1979). For each randomly located plot, physical setting descriptors (aspect, slope angle, topographic position, and elevation), a general stand description (species composition, number of stories, and horizontal spacing), and stand origin (past harvest evidence and regeneration treatment) were recorded. Forest floor characterization included total cover and the proportion of total cover dominated by each char class (unburned, black, grey, or orange colored soils) on a fixed radius plots (1/741 ha). These included new litter (deposition since the fire), old litter (present previous to the fire), humus, brown cubical rotten wood (rotten wood at or above the soil surface), woody debris less than or equal to 7.6 cm in diameter, woody debris greater than 7.6 cm in diameter, rock, and bare mineral soil.

Physical Setting, Fire Weather, and Forest Structure—Fire behavior and burn severity, for the most part, are determined by physical setting (location, topography, juxtaposition, and so forth), fuels (live and dead vegetation), and weather (both short- and long-term). We used the individual fire to reflect the broad scale physical setting. For each burned area we obtained hourly weather observations that occurred during the wildfire. Data from remote automatic weather stations (RAWS) located in the county where each wildfire burned were summarized into daily reports using Fire Family Plus 3.0 (Bradshaw and McCormik 2000). The weather data included relative humidity, maximum temperature, wind speed, and fuel moistures of 1-, 10-, 100-, and 1000-hour fuels. Because the exact day and time a specific plot burned was undetermined, we summarized the weather data to the specific fire. Weather data was unobtainable for some fires located in remote wilderness areas (4 fires).

We used the Forest Vegetation Simulator (FVS) and its Fire and Fuels Extension (FFE) to characterize pre-wildfire forest structure (Wykoff and others 1982, Reinhardt and Crookston 2003, Dixon 2004). Forest structure characteristics included stand density indices, characteristics associated with fire behavior (surface fuels, canopy bulk density, canopy base height), and other miscellaneous stand characteristics (Reinhardt and Crookston 2003). In addition to these FFE-FVS derived forest characteristics we estimated canopy base height directly from our data and described total cover which included canopy overlap as suggested by Crookston and Stage (1999). Also, rather than using quadratic mean diameter (QMD) to describe stem dimensions, we used stem diameter at breast height (d.b.h.) (1.4 m) weighted by basal area[1].

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[1] Basal area weighted diameter breast height (d.b.h.-in) is $\sum$ ((d.b.h.*individual tree basal area (ft^2) * number of trees for each d.b.h. class) divided by $\sum$ (number of trees * individual tree basal area (ft^2).

There are several ways to characterize overstory density such as basal area per unit area, trees per unit area, percent cover, canopy bulk density, relative stand density index, total cubic volume per unit area, and total standing biomass. To avoid collinear variables as predictors, we used canonical correlation for data mining and our expertise to determine which variables had promise for identifying the relation between forest structure and soil burn severity. For density we chose total canopy cover with overlap, for tree size we used basal area weighted d.b.h., average height, and species composition was broadly defined as dry, moist, or cold forest. To describe the forest canopy we used canopy base height (total height minus uncompacted crown length then averaged for plot), and uncompacted crown ratio (fig. 2).

Classifying Burn Severity—Figure 3 illustrates a model we used to develop our soil burn severity classification. The fire literature provided knowledge on fire intensity by describing the heat pulse into the soil (for example, Baker 1929, Debano and others 1998, Hungerford and others 1991, Wells and others 1979). However, the amount of fuel consumed by a fire event also reflects fire intensity. Therefore, we incorporated fire severity into our burn severity classification (for example, Debano and others 1998, Key and Benson 2001, Ryan and Noste 1989) and finally, we included ecological responses

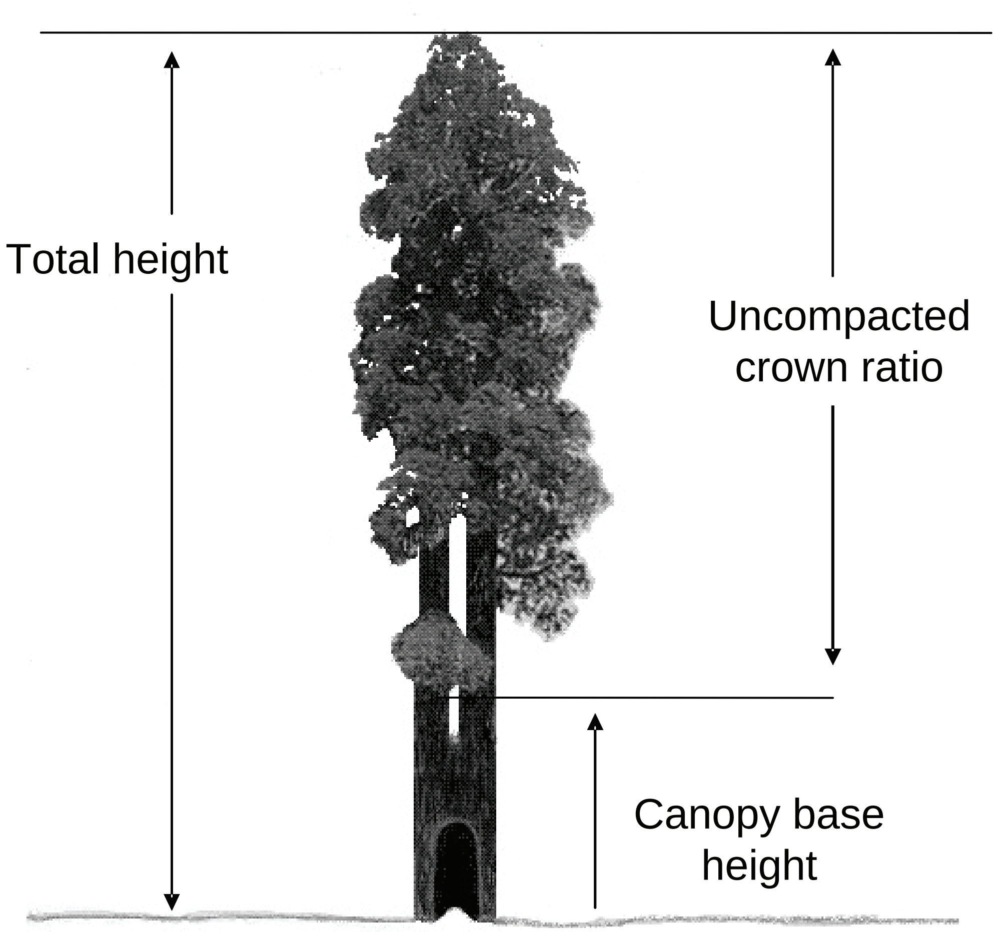

Figure 2—Illustration of how we measured uncompacted crown ratio and canopy base height (total height minus length of uncompacted crown ratio).

that likely occur after a wildfire (for example, changes in wildlife habitat, alterations in soil productivity, changes in soil erosion potential) (Debano and others 1998, Neary and others 1999). As a result our soil burn severity (what is left) classification linked fire intensity, fire severity, and the ecological response (fig. 3).

The classification included six levels of soil burn severity (fig. 4). The factors in the soil burn severity include proportion of litter, mineral soil, and exposed rock present after a fire and their dominant char class, defined as unburned, black char specific to mineral soil, and gray and orange char specific to mineral soil (Wells and others 1979, Ryan and Noste 1989, Debano and others 1998) (fig. 4). The soil burn severity levels included: 1) sites that contained greater than 85 percent litter cover, all char classes, 2) 40 to 85 percent litter cover, all char classes, 3) less than 40 percent litter cover and mineral soil is dominated by black char, 4) less than 40 percent litter cover and mineral soil is dominated by grey or white char, 5) and mineral soil is dominated by black char and no litter cover, and 6) no litter cover and mineral soil is dominated by grey or white char (fig. 4). Wildfires and their "goodness," or lack there of, depends on the values at risk and the biophysical setting and the management

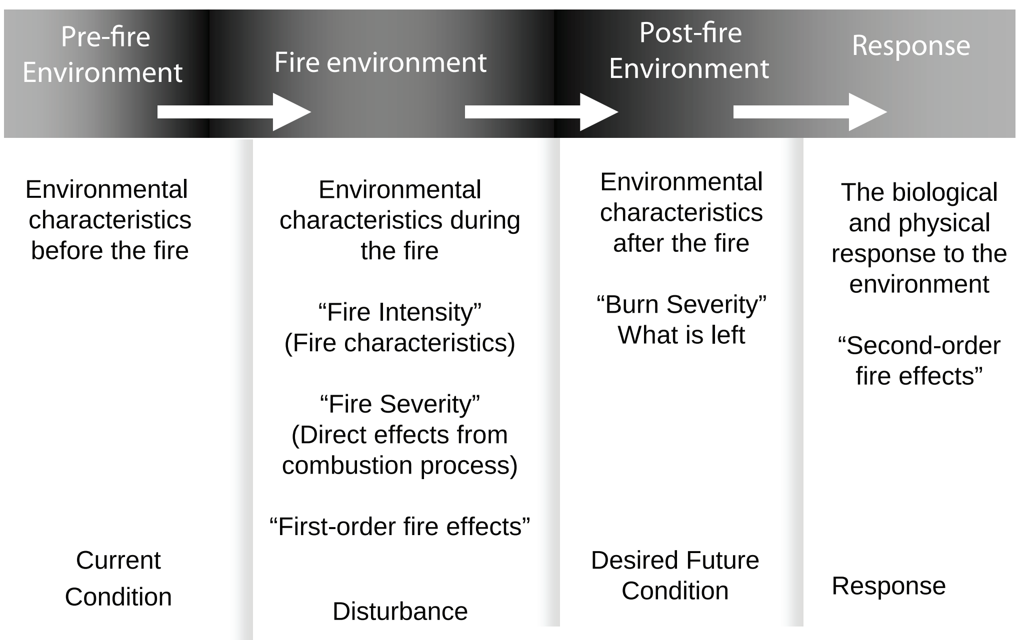

Figure 3—The fire disturbance continuum, of which there are four components, describes the interpretation of different factors involved in wildfires (Jain and others 2004). The first component, the pre-fire environment, includes forest vegetation and state of the environment (moisture levels, amount of biomass, and species composition). This can also be referred to as the current condition just prior to the fire event. The second component, the fire environment, is the environment during the fire event, where fire intensity and fire behavior are characterized in addition to fire severity. Changes to forest components from the fire are also referred to as first-order fire effects. The third component is the environment after the fire is out, referred to as the post-fire environment. This is the environment created by the fire but also is a function of the pre-fire environment and is characterized by what is left after the fire. We refer to this as burn severity. In some cases when fuel treatments are being applied to create a more resilient forest, this could be referred to as the desired condition. The last component is the response, often referred to as second-order fire effects.

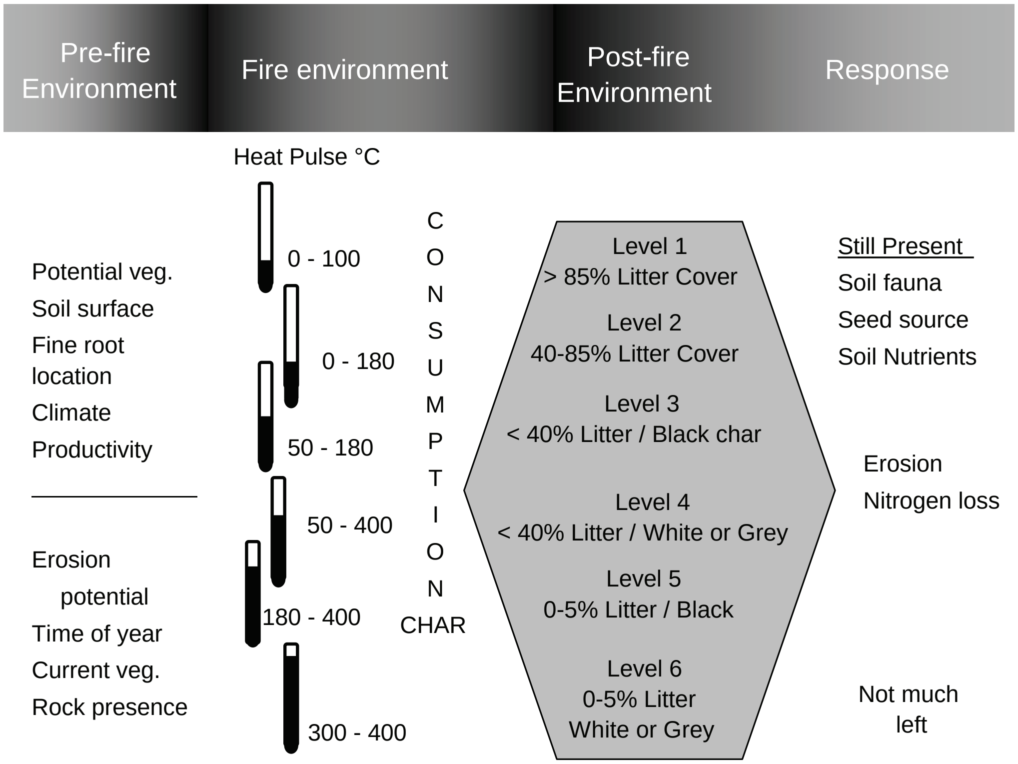

Figure 4—Within the post-fire environment, the soil burn severity classification includes six levels. Going from left to right, a range of temperatures associated with the fire event correspond to the probable indicator of what is left after a fire. For example, to maintain litter cover, the heat pulse into the ground had to be between 0 and 1000 °C. When surface litter is left, often soil fauna are still alive, which often occurs when within a fire severity context, a possible description, is less than 15% of surface litter is consumed. In contrast, by level 6 soil burn severity, the heat pulse into the ground had to exceed 3000 °C in order to create white ash or a grey charred soil appearance (Hungerford and others 1991). The char in each burn severity level refers to the dominant char present after the fire.

objectives for a given setting. Therefore, our six levels of soil burn severity do not depict a value but rather describe a continuum from an unburned forest floor to one in which fire has appreciably altered the physical and biological conditions of the forest floor.

Analysis and Interpreting Results

We combined our six levels of soil burn severity into three levels to ensure our observations were relatively evenly distributed among the different severity classes. Level 2 burn severity (combined level 1 and 2, fig. 4) consisted of areas with greater than 40 percent litter cover ,and the forest floor could vary from unburned to areas exhibiting black char. Level 4 (combined levels 3 and 4, fig. 4) soil burn severity described areas where less than 40 percent litter cover existed and the exposed mineral soil was either black or grey in color. Level 6 soil burn severity (combined levels 5 and 6, fig. 4) described sites where there was minimal litter cover and the exposed mineral soil was black, gray and/or orange colored, or there was an abundance of exposed rock.

We identified relations between forest structure and soil burn severity using a nonparametric classification and regression tree technique (CART) (Breiman and others 1984, Steinberg and Colla 1997). Figure 5 shows a thirteen-outcome classification tree predicting soil burn severity as a function of pre-wildfire forest structure. Outcomes 1 through 13 (shaded) show number of observations correctly classified, total number of observations, and the conditional probability of certainty. Forest characteristics occurring at the top of a classification tree were clearly related to burn severity compared to characteristics that appeared later in the tree. For example, wildfire groups (groups of individual fires) were often the most important in differentiating soil burn severity, followed by uncompacted crown ratio, total cover, and weighted basal area d.b.h. (fig. 5). In addition, the classification tree identified thresholds

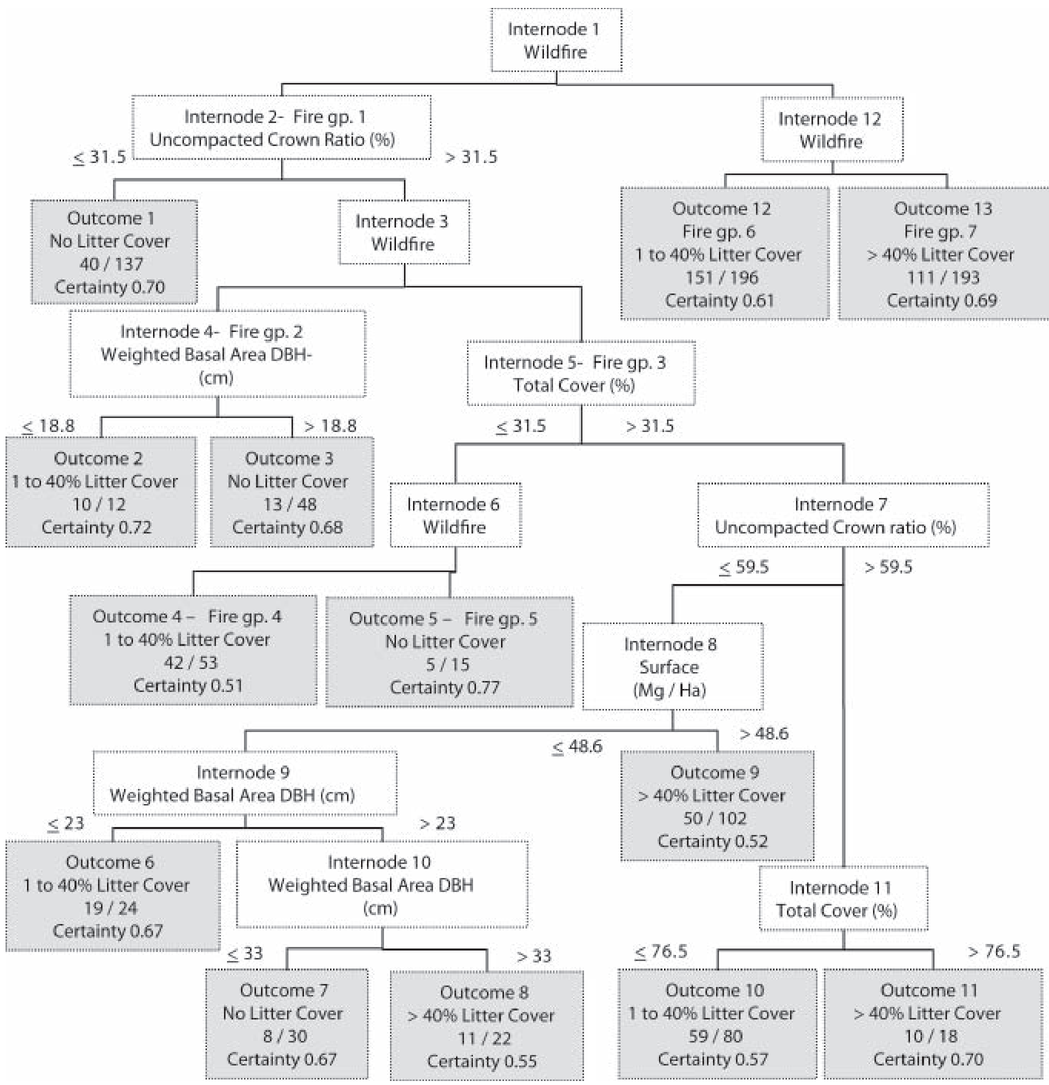

Figure 5—Classification tree for predicting soil burn severity resulting from CART analysis. Shaded areas reflect different predicted outcomes. Each outcome contains the soil burn severity, the number of correctly classified observations versus the total number of observations in the outcome and a conditional probability referred to as "certainty." The internode is where splits occurred based on either fire group or forest structure threshold. Numbers to the left and right of the node indicate the forest structure threshold used in predicting a particular outcome.

at which a forest structure characteristic became related to soil burn severity. In our classification, trees with uncompacted crown ratios ≤31.5 percent were highly related to low litter soil burn severities (level 6, outcome 1) (fig.5). In contrast, trees with uncompacted crown ratios >31.5 percent, differentiated (internode 3) into several outcomes (2 – 8) later in the CART classification. The CART analysis displays conditional probabilities (certainty) of an event happening predicated on earlier classifications. For example, the 0.70 probability of soil burn severity level 6 occurring in outcome 1 is dependent not only if trees have uncompacted crown ratios ≤31.5 percent but also the condition needs to occur within fire group 1 (fig. 5).

Results and Discussion

Our results show that soil burn severity (what is left after a wildfire) is strongly related to general wildfire conditions. That is, we identified seven groups of fires showing similarities when related to soil burn severity (fig. 5). The strength of these relations is exemplified in that fire group 7 only (1 outcome) contained sites with level two soil burn severity (> 40% litter cover, outcome 13). Similarly, fire group 6 only contained sites with level 4 soil burn severity (1 to 40% litter cover, outcome 12). The 56 wildfires in these two groups predominantly burned in the moist and cold forests (figs. 5, 6).

The wildfires in group 3 (outcomes 4 – 11) by far had the greatest diversity in soil burn severity of the wildfires we visited, and the stand structural characteristics often influenced the soil burn severity. Within this fire group total stand cover (internode 5, 31.5%, fig. 5) was an important soil burn severity differentiating characteristic. Stands with the lower canopy covers (≤31.5%) differentiated into two additional fire groups (internode 6, fire groups 4 and 5) and resulted in level 4 (1 to 40% litter cover, outcome 4) and level 6 (no litter cover, outcome 5) soil burn severities (fig. 5). Several of the soil burn severity outcomes (6 – 8) occurring in fire group 3 were related to tree size (weighted d.b.h.) and surface fuel amounts (fig. 5). The wildfires creating these burn severities tended to occur in the dry forests (fig. 6). Also within fire group 3 total cover (internode 11), after uncompacted crown ratio (internode 7), became an important structural element influencing soil burn severity (fig. 5). That is, stands burned in the moist and cold forests with total cover less than 76.5 percent tended to have level 4 (1 to 40% litter cover) soil burn severity and stands having excess of 76.5 percent cover tended to have level 2 soil burn severity (>40% litter cover) (fig. 5). These outcomes (10 and 11) most frequently occurred when wildfires burned the moist and cold forests (figs. 5, 6).

The differentiation of soil burn severity as a result of fire group most likely reflects wildfire characteristics such as fire duration, surface fuel moistures, heat produced, physical setting (for example slope angle, aspect), and geographic location (elevation, landscape position, watershed orientation and juxtaposition). In addition, these results emphasize the importance of observing many wildfires occurring in different years (weather), among many forest types (composition, potential vegetation), and across geographical areas (for example, northern Rocky Mountains, central Rocky Mountains) in order to understand the relation between wildfires and forest structure and how they may determine soil burn severity (Van Mantgem and others 2001).

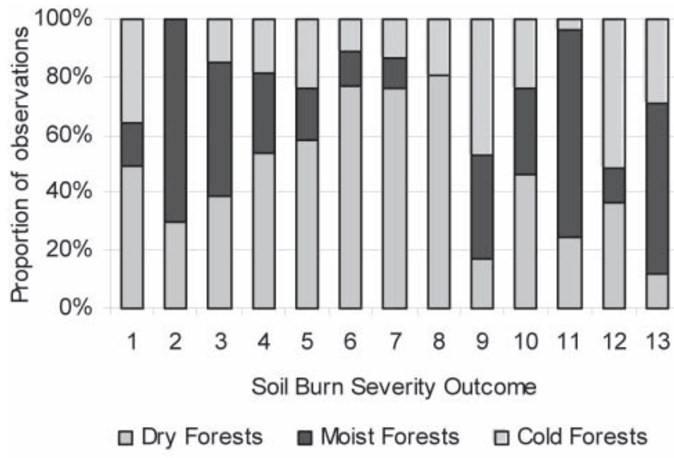

Figure 6—The distribution of forest type within each soil burn severity outcome (see fig. 5). Dry forests are ponderosa pine and/or Douglas-fir cover type. Moist forests are either western hemlock, grand fir, western redcedar, or white fir cover types. Cold forests are subalpine fir and/or lodgepole pine cover types.

Canopy base height, uncompacted crown ratio, and surface fuel conditions most often determine whether a fire will transition from the surface to a crown fire and as a result determine tree burn severity (Scott and Reinhardt 2001, Graham and others 2004, Peterson and others 2005). In contrast, soil burn severity depends on the amount of heat generated on the soil surface, the conduction of heat into the soil layers, and the heat's duration (DeBano and others 1998, Neary and others 1999, Wells and others 1979). These processes are strongly related to the amount of surface fuels, their structure and composition, their moisture content, the pre-fire environment, and the fire environment (fig. 4). Stand characteristics such as tree canopy cover, canopy cover distribution, uncompacted tree crown ratio, and forest composition interact and influence the amount, composition and distribution of live and dead ground-level vegetation (Barnes and others 1998, Oliver and Larson 1990). Therefore, we were not surprised that within a fire group, the most common forest characteristics related to soil burn severity were uncompacted crown ratio, (internodes 2, 7), total cover (internodes 5, 11), tree size (internodes 4, 9, 10), and the amount of surface fuels (internode 8) (fig. 5). Often, these forest characteristics worked in concert and hierarchically to produce a given soil burn severity. For example, for burned over soils to exhibit a level two burn severity (outcome 9) was predicated on sites occurring within fire group 3, trees on the site containing uncompacted crown ratios between 41.5 and 59.6 percent, total canopy cover on the site was less than 31.5 percent, and the surface fuel amounts had to exceed 49.6 Mg ha^{-1} (fig. 5). These results illustrate how overstory characteristics can influence soil burn severity within a group of wildfires and most likely these soil burn severities were related to the amount and condition of ground-level vegetation present when the wildfires burned.

The length of tree crowns in relation to the height of the trees (crown ratio) surprisingly had a strong (differentiated early in the CART analysis) association with soil burn severity, especially with wildfires occurring in group 1 (fig. 5, outcome 1). Fires burning stands with uncompacted crown ratios ≤31.5 percent tended to have no litter cover left after the fires burned, resulting in a level 6 soil burn severity (fig. 5). Many of the stands having this soil burn severity were multi-storied (60 of 127 sites had 3 stories or more) with Douglas-fir trees dominating the dry forests and lodgepole pine trees dominating the cold forests. The trees burned had high canopy base heights (>10 m), the stands averaged 1,900 trees ha^{-1} ($S_{\bar{x}}$ = 196), the mean canopy cover was 40 percent ($S_{\bar{x}}$ = 3) and tree diameter (weighted basal area d.b.h.) was less than 19 cm ($S_{\bar{x}}$ = 1). These results suggest that stands containing trees with short crowns occurring primarily in the cold and dry forests most likely influenced the composition, amount, distribution, structure, and moisture content of the surface fuels. The relatively high tree density may have suppressed surface wind speeds, favoring slow fire spread rates that could have combined with the ground-level vegetation conditions and forest floor surface layers (duff) to favor long duration surface fires. These burning conditions are often attributed to leaving no surface organic matter on a site after a fire and creating black or grey colored mineral soil (Debano and others 1998, Key and Benson 2001, Ryan and Noste 1989).

Stands within fire group 1 and containing trees with uncompacted crown ratios exceeding 31.5 percent differentiated into a multitude of soil burn severities depending on further fire groups, tree diameter, canopy cover, and surface fuel amounts. Within fire group 1 soil burn severity was related to total canopy cover in a subset of wildfires (internode 5, group 3). When burned, the denser stands (cover >76.5%) with crown ratios exceeding 59.5 percent tended to have greater than 40 percent litter cover or level two soil burn severity (outcome 11, fig. 5). Stands exhibiting this soil burn severity usually contained 3 or more canopy layers with mean canopy cover exceeding 90 percent ($S_{\bar{x}}$ = 3) and canopy base heights exceeding 4 m ($S_{\bar{x}}$ = 0.6). This soil burn severity most often occurred within moist forests which tend to have high moisture contents in the surface fuels as a result of the deep and closed canopy conditions. In fact the 1000-hour fuel moisture contents occurring in stands exhibiting this soil burn severity averaged 15.5 percent and were greater than those observed in stands exhibiting the other outcomes (fig. 7). These results indicate that apparently because of the high fuel moistures, moist forests can be relatively resilient to wildfire, even if they contain multiple canopy layers, dense canopy cover, and low canopy base heights.

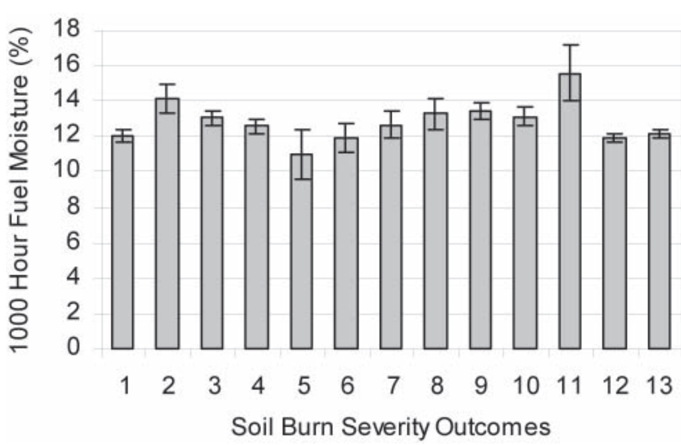

Figure 7—Average fuel moisture and standard errors for the 1000-hour fuels occurring in the stands for each soil burn severity outcome (see fig. 5).

Tree crown ratio appears to influence many stand characteristics that relate to soil burn severity and its influence varies by fire group and canopy cover. After uncompacted crown ratio and canopy cover, the amount of surface fuel becomes influential in determining soil burn severity. However the larger amounts of surface fuels do not readily translate into greater soil burn severity when the forests burned. For example, when wildfires burned stands with crown ratios exceeding 31.5 percent and less than 59.5 percent, canopy cover exceeding 31.5 percent, and containing surface fuels in excess of 48.6 Mg ha^{-1}, level 2 soil burn severity (>40% litter cover) was observed (outcome 9, fig. 5). The moist and cold forests typified this outcome, which historically tend to accumulate large amounts of surface woody debris (80 Mg ha^{-1}, $S_{\bar{x}}$ = 2.5).

After uncompacted crown ratio, canopy cover and the amount of surface fuel, tree size (d.b.h.) becomes a determinant of soil burn severity. The dominance of large trees on a site appear to create conditions that moderate soil burn severity. Soil burn severity level 2 was observed in stands that were dominated by large trees (46 cm, $S_{\bar{x}}$ = 1.0 basal area weighted d.b.h.) even though they contained an average of 40 Mg ha^{-1} ($S_{\bar{x}}$= 0.6) of surface fuels (outcome 8, fig. 5). The canopy cover was moderate (60%, $S_{\bar{x}}$ = 3), as was the canopy base height (7 m, $S_{\bar{x}}$ = 0.6) of stands exhibiting this soil burn severity. This outcome was distributed across the dry forests in strands containing tree densities ranging from 700 to 2,100 trees ha^{-1}. In contrast, level 6 (no litter cover) soil burn severity was observed in predominantly dry forest stands similar to those occurring in outcome 8, except tree diameters were less than or equal to 33 cm. Stands exhibiting this burn severity averaged 28 cm (weighted by basal area) in diameter and contained 1,000 to 2,200 trees ha^{-1}. The mean canopy cover of the stands was 61 percent and the tree canopy base height averaged 4 m ($S_{\bar{x}}$ = 0.5).

These two contrasting soil burn severity outcomes differentiated by tree diameter most likely are related to the tree juxtaposition and variation in density of trees occurring within the stands, especially in ponderosa pine forests, large trees tend to be distributed irregularly often occurring in clumps (Graham and Jain 2005). This irregular horizontal structure would tend to perpetuate variable surface fuel amounts and create a diverse fuel matrix. As a result, surface fires burning fuels in these conditions would most likely result in variable soil burn severities which on the average would be low (level 2). However, small diameter (for example 28 cm) and most likely mid-aged stands, particularly when excluded from fire, tend to develop with more horizontally uniform distributions. As a result, the surface fuels and burning conditions would also be uniform in these stands and may have resulted in surface fires with long residence times.

Small trees (d.b.h.), after uncompacted crown ratio, canopy cover, and the amount of surface fuel were related to level 4 soil burn severity (fig. 5, outcome 6). The dry forest stands dominating this outcome (fig. 5, outcome 6) had 62 percent canopy cover, which was similar to that of the stands occurring in outcomes 7 and 9, but the stands contained more trees (2,000 to 2,800 trees ha^{-1}). Canopy base heights were relatively low (2 m) and average tree height was 13 m ($S_{\bar{x}}$= 1).

The range of soil burn severities occurring among outcomes 6, 7, and 8 illustrate how stand development within dry forests influences soil burn severity. The small diameter young forests when burned tended to create level 4 soil burn severities (outcome 6), the stands with mid-sized and likely mid-aged trees when burned tended to create level 6 soil burn severities (outcome 7, fig. 5), and when stands containing large and old trees burned, level 2 soil burn severities were created (outcome 8, fig. 5).

In fire group 2, which is a subset of group 1 fires, tree size was second only to uncompacted crown ratio in explaining soil burn severity. Again, diameter most likely reflects a developmental stage of the stands exhibiting the two contrasting burn severities. Stands with the smaller and younger trees (<18.8 cm, weighted basal area d.b.h.) had level 4 burn severity compared to the stands containing the mid-aged and larger trees (>18.8 cm weighted basal area d.b.h.) which exhibited level 6 burn severity (no litter). These findings were similar to those illustrated in outcomes 6 and 7 except these outcomes occurred in fire group 2 and outcomes 6 and 7 occurred in fire group 3 (fig. 5). The moisture content of the 1000-hour fuels in stands occurring in outcome 2 was 14 percent ($S_{\bar{x}}$ = 1) and 11 percent ($S_{\bar{x}}$ =1) for the 1000-hour fuels within stands occurring in outcome 3.

Thinned stands, plantations, and others exhibiting management typified stands in outcomes 2 and 6. The forest floor conditions of stands in these outcomes most likely resembled those associated with stand initiation structural stages. These early structural stages frequently contain moist and robust layers of ground-level vegetation. Because these stands were managed, the surface fuel matrix was modified through slash disposal and site preparation activities resulting in a discontinuous fuel bed. Particularly, in the cold and moist forests, crown fires would burn around these areas and most often there was evidence that firebrands landed in these stands but the surface fuel conditions prevented sufficient fire from developing that could create a smoldering fire. Therefore, these results indicate that high stand densities and low canopy base heights do not necessarily lead to severely burned soils and other factors such as developmental stage may also influence soil burn severity.

After uncompacted crown ratio (>31.5%) and total canopy cover (<31.5%) the fire setting (fire group) became an important predictor of soil burn severity (fig. 5). Two fire groups differentiated, one expressing level 4 soil burn severity (outcome 4, fire group 4) and one expressing level 6 soil burn severity (outcome 5, fire group 5). Both outcomes had similar representation from cold, moist and dry forests (fig. 6) and the stand densities of both were low (292 trees ha^{-1} for outcome 4 and 312 trees ha^{-1} for outcome 5) when compared to stand densities occurring in the other outcomes. Also, for both outcomes canopy base heights were near 6 m and the uncompacted crown ratios for both were above 60 percent. The greatest difference in the stands occurring in the two outcomes was the setting (for example topography, geographic location, watershed juxtaposition and so forth) in which they occurred. Outcome 5 consisted of observations from the Hayman and Missionary Ridge fires in Colorado and the Ninemile fire in Missoula County, Montana. Outcome 4 included observations from the Alpine, Bear, and Blodget fires in Ravalli County, Montana and the Flagtale fire in Grant County, Oregon. The stands burned by wildfires in outcome 4 also had higher 1000-hr fuel moistures (12.5%) than stands burned by the fires in outcome 5 (11%) (fig. 7). In addition, the average wind speeds occurring during the fires in outcome 5 tended to be higher (7 to 8 miles hour^{-1}) when compared to the winds blowing during outcome 4 fires (4 miles hour^{-1}). The different burning conditions (for example fuel moisture, wind speed, location, and so forth) exemplified in these two outcomes probably had a greater influence on soil burn severity than forest structure, given that both outcomes had very similar structural characteristics.

There are several factors (for example, weather, type of vegetation, fuel moisture, atmospheric stability, physical setting, ladder fuels, surface fuels) that influence fire behavior and burn severity, and forest structure is only one (Agee 1996, Graham and others 2004). Therefore, we did not expect forest structure to fully explain all of the variation present in soil burn severity after a wildfire. However, through our study and the analysis we performed, we were able predict soil burn severity as a function of pre-wildfire forest structure with probabilities far greater than what would have occurred randomly. These variables were not only hierarchally related to soil burn severity, but together they very readily predicted three levels of soil burn severities. Because we identified three levels of soil burn severity, a random probability of a given soil burn severity occurring would be 0.33. Therefore, any probability exceeding 0.33 of the complete CART tree correctly classifying a particular soil burn severity indicates the addition of forest structural characteristics were significantly related to soil burn severity. The variables, in order of importance, fire group, uncompacted crown ratio, weighted basal area d.b.h., total cover, and surface fuel amounts classified level 2 soil burn severity (>40% litter cover) with a 0.46 probability, level 4 soil burn severity (1 to 40% litter cover) with a 0.40 probability, and level 6 (no litter cover) soil burn severity with a 0.57 probability.

Conclusion

Undoubtedly intense fire behavior is a primary concern for forest management throughout the western United States and fuel treatments to modify this fire behavior are a primary concern (Graham and other 2004). However, in most circumstances what a fire leaves behind in terms of soils, homes, and trees is as important, if not more important than fire behavior. Therefore, fuel treatments need to be designed and implemented as to modify burn severity and the traditional thinned forest with high canopy base heights may not result in the desired burn severity.

One size does not fit all. Therefore, we would suggest that fuel treatments be designed to consider burn severity as well as fire behavior. In particular, biophysical setting (fire group, forest type, locale, potential vegetation type, and so forth) needs to provide context for planned fuel treatments. Secondly, tree canopy base height (reflected in uncompacted crown ratio) needs to be considered when designing fuel treatments, although high canopy base heights do not always reduce soil burn severity. Similarly, reducing total forest cover does not necessarily reduce soil burn severity; rather its interactions with the biophysical setting, canopy base height, and surface fuel amounts and conditions most likely determine soil burn severity. The last characteristics that we identified as having a relation with soil burn severity, were tree diameter and surface fuel amounts.

The robust data we accumulated from wildfires that burned throughout the western United States in recent years did not greatly simplify our understanding of the relations between forest structure and soil burn severity. Nevertheless, we did identify several interactions between forest characteristics and soil burn severity that have fuel treatment management applications. A significant factor of this work is the estimate of the certainty a forest structure (fuel treatment) will have in modifying soil burn severity. The conditional probabilities (certainty) we identified of forest structure or fire setting (fire group) influencing soil burn severity always exceeded 0.50 and occasionally exceeded 0.75 (fig. 5). In addition, the approach we took in identifying the relations between forest structure and burn severity, and the level of certainty we provided, was conditional on the circumstances in which the forest characteristic occurred. This kind of information will be of value when communicating the importance forest structure (fuel treatments) has on determining the aftermath of wildfires. This paper and the analysis and results we reported are a continuation of our work in understanding how forest structure interacts with wildfires, their biophysical setting, and burning conditions to create a particular burn severity.

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